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ENVIRONMENTAL RESEARCH

BIODEPOSITION AND UPTAKE OF POLYCHLORINATED BIPHENYLS AND CADMIUM BY THE ZEBRA MUSSEL (*Dreissena polymorpha*)

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OF POLYCHLORINATED BIPHENYLS
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(Dreissena polymorpha)

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FINAL REPORT

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ABSTRACT

BIODEPOSITION AND UPTAKE OF POLYCHLORINATED BIPHENYLS AND CADMIUM BY THE ZEBRA MUSSEL (Dreissena polymorpha)

Biodeposition and accumulation of polychlorinated biphenyls (PCBs) and cadmium (Cd) by zebra mussels was investigated in the western basin of Lake Erie and Hamilton Harbour during the summer and fall of 1992. Results indicated that mussels significantly ($p<0.05$) increase the deposition rate of suspended material, organic matter, PCBs, and Cd from the water column through biodeposition processes. Levels of suspended inorganic and organic matter, and chlorophyll a were significant factors in models predicting biodeposition rates of suspended matter, PCBs, and Cd. The biodeposition rate of suspended material and contaminants converted to a per unit area basis was significantly ($p<0.001$) greater than that for natural sedimentation.

Zebra mussels transplanted from Port Stanley, Ontario, into the western basin of Lake Erie had significantly ($p<0.05$) greater levels of PCBs and Cd after their exposure than did the initial Port Stanley mussel samples, and had similar to or greater levels than *in situ* mussels in the western basin sites. Levels of suspended inorganic and organic matter, and chlorophyll a were significant factors in models predicting accumulation rates of PCBs and Cd in mussel tissue. Tissue accumulation rates of PCBs and Cd converted to a per unit area basis were very high.

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CHAPTER ONE:

GENERAL INTRODUCTION

1.1 BRIEF REVIEW OF POLYCHLORINATED BIPHENYLS

Polychlorinated biphenyls (PCBs) are a group of compounds belonging to a family of chemicals called the chlorinated hydrocarbons. The structure of PCBs is based upon a biphenyl molecule (Fig. 1.1) which can be chlorinated at some or all positions around the rings, allowing for some 209 possible combinations, or isomers, of the molecule.

PCBs are a synthetic group of chemicals originally manufactured for several purposes, but most importantly as a dielectric fluid in power transformers and capacitors (McDonald and Tourangeau 1986). They are an extremely inert group of compounds, thus making them useful for such purposes. However, PCBs were later found to be extremely persistent in the environment upon careless discharge, as well as being able to bioaccumulate in the food web. In Canada in 1977, PCBs were banned from production and presently are found only in transformers and capacitors which were installed before this date (Strachan 1988).

The toxicity of specific PCBs varies with molecular structure. The toxicity of polychlorinated dibenzo-*p*-dioxins (PCDD) and dibenzofurans (PCDF) is well known (Fletcher and McKay 1993; Tanabe 1989), and within this group, 2,3,7,8 tetrachlorodibenzo-*p*-dioxin (2,3,7,8 T_4CDD) is considered the most toxic (Tanabe et al. 1987; Fletcher and McKay 1993). Chlorine atom positioning on the 2,3,7,8 T_4CDD molecule sterically allows for the biphenyl rings to be in the same plane, or coplanar, which is considered biochemically important from the standpoint of toxicity (Tanabe et al. 1987). Of the 209 possible congeners of PCBs, there are 20 which can have a coplanar structure, and therefore have toxic effects similar to 2,3,7,8 T_4CDD. Toxic effects include body weight loss, thymic atrophy, dermal disorder, hepatic damage, teratogenicity, reproductive toxicity, and immunotoxicity (Tanabe 1989). Although the long term effects of trace level exposure of these chemicals on humans and animals is unclear (Tanabe et al. 1987), the above studies strongly suggest the potential toxic importance of coplanar PCBs.

1.2 ORGANIC CONTAMINANT CYCLING

The cycling of organic contaminants in the aquatic ecosystem is influenced by chemical, physical, and biological processes (Allan 1986). The input of contaminants into the aquatic environment can be from point source origins (eg. industrial inputs), and from non-point source origins (e.g. atmospheric deposition) (Baker and Eisenreich 1989). Once in the water, these chemicals tend to rapidly associate with suspended particulates, and in so doing much of the cycling of these contaminants is intimately linked with particulate dynamics throughout the water column (Baker et al. 1991). Through sedimentation of these particulates the associated contaminants are deposited on the bottom, where in turn they can be resuspended through physical and biological means, or effectively removed from the system through sediment burial.

Sorption of organic contaminants to suspended material is based on the specific partitioning coefficient of the chemical (K_d) between suspended material and water (Allan 1986). The K_d is characteristic of the contaminant and the sediment properties (Allan 1986). In particular, organic contaminants preferentially sorb to organic material (dissolved and particulate) in the water column because of their lipophilic nature (Servos and Muir 1989). The cycling of organic contaminants is therefore ultimately linked with that of organic matter (Baker et al. 1991; Baker and Eisenreich 1989).

Organic matter cycling is initiated in the epilimnion through formation of particulates from primary production. Organic contaminants free in the water can sorb to these surfaces or can be actively taken up by the living cells. These particulates produce dissolved organic matter (DOM) through excretion products, or through breakdown, which can also act as important substrates for contaminant sorption. DOM and particulate organic matter (POM) in the epilimnion can coagulate, or be packaged as faeces by zooplankton, facilitating settlement as masses. Such settlement is an effective mechanism of removal of these chemicals from the water column because of the association of contaminants with DOM. These settling particles can further scavenge organic contaminants throughout water column transport. Alternatively, upon degradation sorbed contaminants can be released back into the water, or can remain associated with residue particulates (Baker and Eisenreich 1989; Fletcher and McKay 1993).

In deeper bodies of water where stratification occurs, a layer of high suspended particulate concentration, called the benthic nepheloid layer (BNL), can extend from the bottom upwards for several metres (Baker and Eisenreich 1989; Baker et al. 1991; Allan 1986). This layer is created by resuspension processes caused by internal currents as well as bioturbation by benthic organisms (Fletcher and McKay 1993). Associated with the high suspended particulate concentrations in the BNL are the sorbed contaminants, which are therefore found at high concentrations in this region. During fall turnover this contaminant pool is mixed throughout the water column, and thus the overall effect of the BNL is to allow for a much longer residence time of organic contaminants in the water column than would be afforded by sedimentation processes alone (Baker and Eisenreich 1989). This in turn can influence exposure time to aquatic biota.

1.3 ORGANIC CONTAMINANT BIOAVAILABILITY

Throughout the contaminant cycling process, aquatic organisms can be exposed to organic contaminants at several points. The process of uptake of these chemicals is dependent upon the bioavailability of the chemical. Bioavailability of a contaminant is defined as external availability of the chemical to an organism (Schrap 1991). There are two general uptake routes; through respiratory surfaces such as gills, and through the gastro-intestinal tract (G.I. tract) (Schrap 1991). The extent of uptake via either of these routes is dependent upon the nature of the organism, the chemical, and the substance to which the contaminant is sorbed (Knezovich et al. 1987).

The uptake of a contaminant by a organism can be affected by the species' physiology, ecological habitats, and behaviour (Knezovich et al. 1987). For example, infaunal species such as oligochaetes may accumulate contaminants through the cuticle as well as through the gut upon exposure to contaminated sediments and interstitial water (Knezovich et al. 1987; Oliver 1984). Absorption of dissolved chemicals in the water column over respiratory surfaces is an important route of exposure for fish. Filter feeders and deposit feeders, such as mussels, may be particularly vulnerable to uptake of contaminants through the G.I. tract since these organisms actively feed on settling and settled contaminated detritus (Knezovich et al. 1987; Schrap 1991). In addition, the organism size is important, i.e. smaller sized individuals of the same species tend to accumulate significantly greater amounts of contaminants per unit weight than do larger animals (Oliver 1984), probably due to a greater surface area to volume ratio in smaller individuals.

Chemical properties also play an important role in Bioavailability. For organic contaminants the affinity for organic matter, or lipophilicy, is a measure of potential uptake into the lipids of animals. The K_{ow}, or octanol/water partition coefficient of a chemical, is a measure of the distribution of a given organic chemical between octanol and water. The higher the K_{ow} the greater is the lipophilic nature of the chemical. The K_{ow} then should theoretically predict the extent of uptake of a contaminant into the lipids of an organism relative to the water concentration. Several studies have confirmed this to some extent, namely tissue concentration was positively related to log K_{ow}, but then negatively related above a critical log K_{ow} value (Oliver 1984; Servos and Muir 1992). This was attributed to possibly the larger size of the molecule (increased chlorination) and/or the extreme lipophilicy, preventing the molecule from passing through biological membranes due to size and the preference for the sorbing agent, respectively.

The nature of the substance to which the organic contaminant is associated is also important in determining the contaminant's Bioavailability. Organic particulates are important sorbing agents for contaminants. These particulates are used as food sources for several species, particularly scavengers and filter feeders, and therefore are a potential uptake route for such animals (Servos et al. 1992; Schrap 1991). McCarthy (1983) concluded that polyaromatic hydrocarbons adsorbed to particulates which could be filtered by Daphnia magna consisted of some of the bioavailable fraction. Servos et al. (1992) concluded that for some dioxins, detrital food chain transfer was the most important route of uptake for various test organisms. Alternatively, sorption of organic contaminants to DOC and POC in the water column reduces the availability of the contaminant when uptake through respiratory membranes is considered. Servos et al. (1992) and Servos and Muir (1989) attributed the reduced availability to the large size of the contaminant-sorbing agent complex which was unable to pass through the membranes. Uptake through respiration is most efficient when the chemical is 'free', or truly dissolved in the water column, thus enabling transfer over the membrane.

1.4 BRIEF REVIEW OF CADMIUM

Cadmium (Cd) is a heavy metal which is closely related to zinc and mercury in its chemical properties (NRCC-ACSCEQ 1979). In its most stable state in the aquatic environment, Cd is found as a divalent atom (Cd^{2+}). Cd is chiefly found in sulfide minerals and is removed from rock as a by-product of zinc ore processing (NRCC-ACSEQ 1979). It is also a by-product of the refining of other metals (Moore and Ramamoorthy 1984).

Cadmium has been subject to increasing industrial use over the past several decades. Some of the main uses of Cd include electroplating, pigments in paints and plastics, stabilizers in plastics such as polyvinyl chloride (PVC), and in rechargeable nickel-cadmium batteries (NRCC-ACSCEQ 1979; Moore and Ramamoorthy 1984). Extraction of Cd for industrial uses is usually combined with zinc extraction processes, where Cd as a by-product is then collected (NRCC-ACSCEQ 1979).

Because Cd is often a by-product of ore processing, much of this element has made its way into the aquatic environment through mining activity (NRCC-ACSCEQ 1979), and it is also subject to mobilization from rock through acid rain-induced erosion. Once mobilized the chemical can potentially accumulate in humans and other biota through consumption of aquatic life. Health effects of Cd on humans are not well understood, but it is evident from the literature that they may be both acute and chronic. Symptoms of acute Cd toxicity include: abdominal pain, nausea, vomiting, diarrhoea, headache, and vertigo, and lethal effects stemming from liver and kidney dysfunction (Yasumura et al. 1980; Stephenson 1986). prolonged chronic exposure to Cd also caused renal and liver disease as well as proteinura in Swedish workers exposed to Cd oxide dust (Friberg 1949, cited in Stephenson 1986) and osteomalacia and kidney disease in people exposed to contaminated water in Japan (Yasumura et al. 1980).

1.5 INORGANIC CONTAMINANT CYCLING

Inorganic contaminants, like organics, are found associated with various particulate and dissolved fractions in the water column. Therefore, both types of contaminants share similarities in cycling processes that involve these fractions. Some differences do exist, however. Metals have been shown to also have a high affinity for organic matter (Morfett et al. 1988; Allan 1986), but are strongly associated with manganese and iron oxides as well. These oxides are subject to redox transformations, which in turn affect the adsorption/desorption processes of the associated metal contaminants (Morfett et al. 1988). In anoxic or reducing sections of the water column, these oxides are reduced to soluble forms of Mn(II) and Fe(II), and release associated metals into the water above the sediments (Tessier et al. 1993; Morfett et al. 1988). Upon being transported into areas of the water column where Mn(II) and Fe(II) can be reoxidized, free metal ions can be scavenged by these compounds. Under extremely reduced conditions, the free metal ions can associate with iron sulfides and be precipitated into the sediments (Luoma 1989; Morfett et al. 1988). Sigg et al. (1987) suggest that in the deep part of their study lake, redox driven cycling of Mn and Fe oxides and associated metals is an important removal mechanism of these contaminants.

1.6 INORGANIC CONTAMINANT BIOAVAILABILITY

As with organic contaminants, uptake of metals over respiratory surfaces and the G.I tract of an organism constitutes the most important entry routes of these elements. Metals in solution and bound to particulates (food) are the important phases. Metals in either phase are further partitioned among different ligands, based on ligand concentration and the strength of the metal-ligand bond (Luoma 1983). Therefore organisms are exposed to several forms of the metal, each varying in availability to the animal.

In solution, a metal is most bioavailable in its free state. When complexation with dissolved inorganic or organic materials occurs, the availability generally decreases (Luoma 1983), presumably because of the inability of the complex to pass over the respiratory membranes. Generally the best relationship between metals in the environment and their uptake by the organism is seen when metals in solution are considered (Tessier et al. 1993). Alternatively, metals in sediments are found in several forms (metal-ligand) which can pass through the G.I. tract upon ingestion by benthic animals. Predictions are not as easily made with particulate phase uptake of the contaminant as with solution phase uptake because these partitioning processes in sediments are poorly understood, and vary considerably due to varying sediment compositions (Luoma 1989).

Correlations between total metal concentration in sediments and levels in the animal do not have good predictive value (Allan 1986). In an attempt to determine availability of metals from sediments, a sequential extraction technique (S.E.T.) was developed to partition metals among different sediment types (Tessier et al. 1984). Each extraction could then be compared with metal levels in the test organism to determine the most important fraction. Unfortunately, this process does not completely select for specific sediment types, but rather each fraction is only defined by the extraction procedure, which approximates these types (Luoma 1989). However, Tessier et al. (1984) found an effect of sedimentary iron oxide, where FeO_2 sorbed metals were less available to Elliptio complanata. Harvey and Luoma (1985) observed a decrease in Cd accumulation for Macoma balthica when organic matter was stripped from the sediments; they attributed the decrease in Cd accumulation to the removal of the organic source of Cd exposure. As well, accumulation of Cd was non-existent when FeO_2 was the dominant component of the sediments. Alternatively, Tessier et al. (1993) found no correlation between Cd levels in Anodonta grandis and extractions of Cd from sediments which were normalized with respect to FeO_2 and organic matter. Therefore predictions of Bioavailability of metals from sediment appear difficult to do, yet sediments as a source of contamination are quite important.

1.7 UPTAKE AND DEPOSITION OF CONTAMINANTS BY ZEBRA MUSSELS

The recent invasion of the Great Lakes by the zebra mussel (Dreissena polymorpha) is proving to be of major ecological significance. This includes the important role that zebra mussels may now play in contaminant dynamics in these bodies of water. Two mechanisms by which zebra mussels may affect contaminant dynamics are: increasing the biodeposition rates of contaminants bound to suspended material, and accumulating contaminants in their tissues so that the mussels themselves may act as a pool for the storage of contaminants during cycling (Fig. 1.2), and for movement into higher trophic levels through predation.

Zebra mussel filtration rate on a per unit weight basis is comparable to that of native bivalves (Kryger and Riisgard 1988), but when the much higher densities of these mussels relative to other bivalve species are considered, this aspect can be ecologically important. Associated with filtration is the resulting deposition of faeces and pseudofaeces by the mussels (Haven and Morales-Alamo 1966). The enhanced filtering capacity of large mussel populations has been shown to be potentially important in the restoration of eutrophic bodies of water (Reeders et al. 1989). It has been suggested that zebra mussels are altering the chlorophyll a (chl a) levels in Lake Erie because of their filtration activities, especially in the water column above mussel beds (MacIsaac et al. 1992). Stanczykowska et al. (1976) concluded that faecal deposition by zebra mussels constituted about 13% of the yearly sedimentation in the pelagic zone of a European lake. Biodeposition by other mollusc species has also been shown to increase the natural sedimentation rate of suspended particulates (Kautsky and Evans 1987; Jaramillo et al.

1992). Reeders and Bij de Vaate (1992) have shown that biodepositional material is more contaminated with various pollutants than suspended matter. Therefore, zebra mussels could be potentially altering contaminant dynamics by increasing the sedimentation rate of contaminated suspended material.

Deposition by zebra mussels will be influenced by environmental factors such as overall suspended material concentration, food concentration, and inorganic matter concentration, because these factors influence filtration rates (Sprung and Rose 1988; Stanczykowska et al. 1975). **Therefore, the first objective (chapter two) of this study was to assess the ability of zebra mussels to remove particulate-bound total PCBs (t-PCBs) and Cd from the water column through biodeposition in faeces and pseudofaeces, and to determine significant factors influencing biodeposition.**

Zebra mussels have been shown to accumulate high levels of contaminants in European waters (Reeders and Bij de Vaate 1992) and in the Great Lakes (Secor et al. 1993), often to levels greatly exceeding those found in native bivalves (Brieger and Hunter 1993). This, in conjunction with the species' very high densities in the Great Lakes, may enable the animals themselves to be an important point in contaminant dynamics through uptake of these chemicals into their tissues. Brieger and Hunter (1993) have suggested that zebra mussels may be altering contaminant movement by this means in Lake St. Clair.

Accumulation of PCBs and Cd will be influenced by the Bioavailability of the contaminant and the physiology and behaviour of the animal. Zebra mussels, being suspension feeders, can accumulate contaminants from the water (over respiratory membranes) and from food. Brieger and Hunter (1993) reported that food is a particularly important route of accumulation of PCB congeners in zebra mussels. Therefore, the second objective (chapter three) of this study was to assess the ability of zebra mussels to accumulate t-PCBs and Cd in their tissues, and to assess the potential impact of zebra mussels on contaminant dynamics as a consequence of this accumulation.

PCBs were chosen for this study as a representative organic contaminant of concern. These chemicals are persistent and toxic, as well as being able to accumulate to high levels in aquatic organisms. In addition, they are generally found at much higher levels than the related chlorinated dioxin and furan group of chemicals, which enable the analytical processes involved in PCB detection technically easier and less expensive than that for dioxins and furans.

Cadmium was chosen for this study as a representative inorganic contaminant of concern. Like PCBs, Cd is highly toxic and has the ability to accumulate in aquatic biota. Unlike some other metals, Cd is biologically non-essential (Förstner and Wittmann 1981) and is found in relatively high levels in the waters of the Great Lakes, thus facilitating detection.

The thesis is divided into three chapters. Chapter two addresses objective one. Chapter three addresses objective two, and also includes a section of overall conclusions and the potential ecological implications of these conclusions. Hypotheses and predictions relating to each objective are developed in the introduction of each chapter.

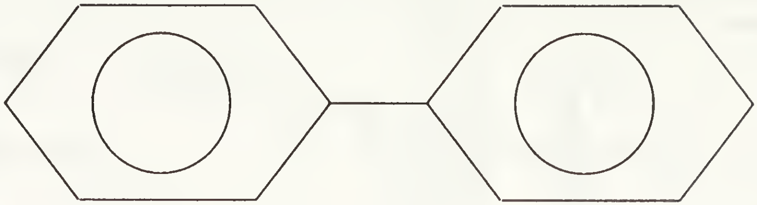
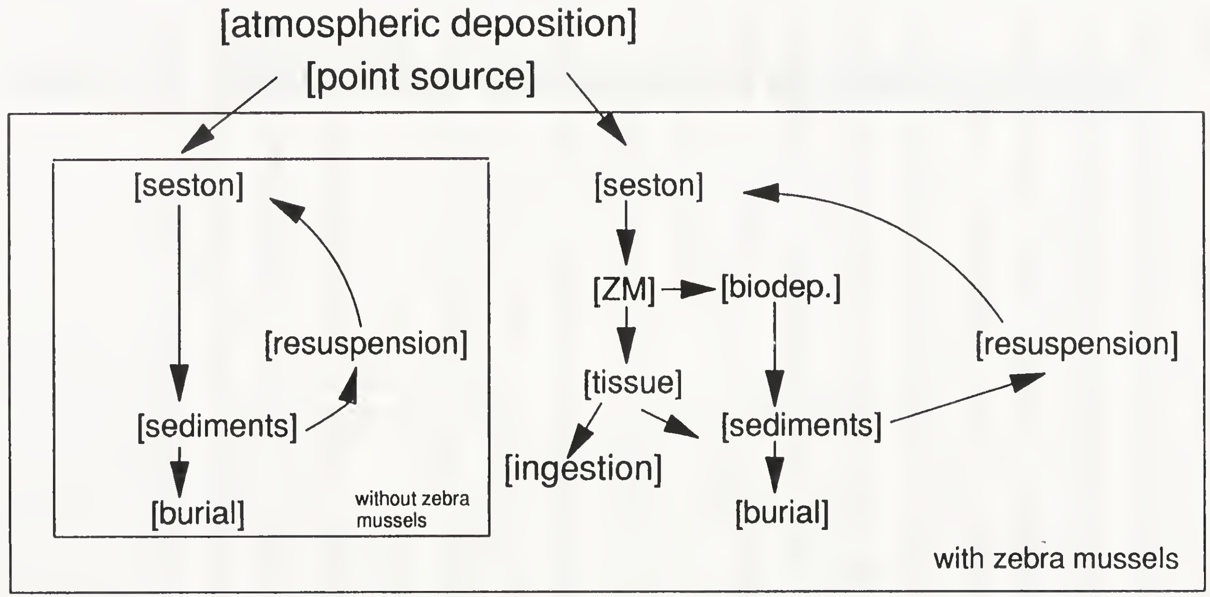


Figure 1.1. Diagram of the base structure of the biphenyl molecule.



[ ] = contaminant concentration
ZM = zebra mussel

Figure 1.2. Schematic representation of the potential effects of zebra mussels on contaminant dynamics.

CHAPTER TWO:

DEPOSITION OF PCBs AND Cd BY ZEBRA MUSSELS (Dreissena polymorpha) IN WESTERN LAKE ERIE AND HAMILTON HARBOUR

2.1 INTRODUCTION

Zebra mussels may alter contaminant movements in the water column through their biodeposition processes. The first overall objective of this study was to assess the ability of zebra mussels to remove particulate bound PCBs and Cd from the water column through biodeposition. Within this objective are two hypotheses: 1) zebra mussels significantly change the deposition rate of PCBs and Cd over that of natural sedimentation; and 2) there are significant relationships between total suspended solid concentration, and food quality factors (e.g. organic/inorganic matter levels in suspended particulates, chl a) and the biodeposition rate of PCBs and Cd by zebra mussels.

It has been shown that biodeposition by bivalves is similar to, or greater than, natural sedimentation (Kaustsky and Evans 1987; Jaramillo et al. 1992). Significant ecological impacts due to zebra mussel filtration and biodeposition in European waters have also been observed (Stanczykowska et al. 1975; Reeders et al. 1989). Reeders and Bij de Vaate (1992) have shown zebra mussel biodeposits to be more contaminated than suspended material. Given these results, the prediction of the first hypothesis is that zebra mussels will significantly increase the deposition rate of PCBs and Cd over that of natural sedimentation.

Food concentration and food quality are environmental factors which have been shown to influence the biodeposition rate of suspended material by bivalves. The process of psuedofaeces formation is an effective means for bivalves to deal with overall high suspended loads of particulates and/or food, as well as high proportions of inorganic suspended particulates (Sprung and Rose 1988). Haven and Morales-Alamo (1966) saw positive correlations between inorganic seston and pseudofaecal production. Reeders and Bij de Vaate (1992) observed increased biodeposition by zebra mussels with increasing overall suspended dry matter content. Similar results were seen by Haven and Morales-Alamo (1966). Increased biodeposition with increasing food concentration has also been observed in various mussel species by Tenore and Dunstan (1973) and Tsuchiya (1980). Given the findings of these studies, the present study will attempt to model the biodeposition of contaminants by zebra mussels. If biodeposits and suspended material have a similar concentration of contaminants, the prediction of the second hypothesis is that biodeposition of PCBs and Cd by zebra mussels will be positively correlated with total suspended solid concentration, food quantity (measured as overall organic suspended matter and chl a), and suspended inorganic matter content.

2.2 METHODS AND MATERIALS

2.2.1 Site Location and Sediment Trap Deployment

In August, 1992, studies were conducted in the western basin of Lake Erie to determine the effects of zebra mussel biodeposition on PCB and Cd sedimentation. Five locations were chosen for this purpose (Fig. 2.1, Table 2.1). Locations with Coast Guard buoys (with the exception of Point Pelee, Fig. 2.1) were selected to facilitate relocation, as well as to address objectives in chapter 3 of the study. At each location, two depths were used; one near the top of the water column (3 m below the surface) and one near the bottom of the water column (1.5 m above the bottom, 8-12 m below the surface). Therefore, there were 10 sites (2 depths per location) used in the study. Locations were selected so that variability in environmental factors (e.g. chl a, seston) and contaminants (Cd and PCBs) between sites was maximized while contaminant levels were high enough for detection. A gradient in environmental factors (e.g. chlorophyll a, Glooschenko et al. 1974) and contaminants (Farara and Burt 1993) exists in the basin. Different depths were chosen as sites because variation in environmental factors and contaminants within the water column has been observed in other studies (e.g. chlorophyll a and particulate matter, Jaramillo et al. 1992; Cd and PCBs, Dobson 1991). Open water locations were chosen to reduce problems of high resuspension of bottom sediments associated with shallow inshore locations (discussed later in this section).

Sediment traps were used to determine the effects of zebra mussel biodeposition on PCB and Cd sedimentation. The trap design is shown in Fig. 2.2. The efficiency of a sediment trap in trapping settling particles is dependent upon a trap's aspect ratio (height:diameter) and the absolute diameter of the opening (Roff and Hopcroft 1985). Specifically, the aspect ratio and the Reynolds number (Re) in combination for a given trap design are the most reliable means of determining the efficiency of a sedimentation trap in trapping settling material (Lau 1979). The Re, in the case of sediment traps, is equal to the inner trap diameter multiplied by the current velocity divided by the kinematic viscosity of the water (0.01 Stokes at 20°C, Hawley 1988). The principle behind sediment trap design is to allow for water exchange between the inside and outside of the trap, while also maintaining a non-turbulent bottom layer of water in the trap so that settled material will not resuspend. Lau (1979) observed that traps with an aspect ratio of 5:1 and a Re $<$ 7000 did not readily resuspend settled material.

In the present study, the aspect ratio of the sedimentation traps was 5.3:1. It is difficult to calculate the Re for these traps since under field conditions the current velocity in the western basin of Lake Erie is highly variable. For this reason, few studies have studied the current patterns in the western basin. Current speeds are usually less than 20 cm/s in the Great Lakes (Roff and Hopcroft 1985). MacIsaac et al. (1992) measured current speeds at their study site in the western basin of Lake Erie of 0.9 to 3.4 cm/s, but suggest that these values were probably recorded during calm periods. If the value of 3.4 cm/s is doubled to account for any potential differences in currents between the present study and those from MacIsaac et al. (1992), then my traps would have a Re of 6434, which is under the limit suggested by Lau (1979) ($<$ 7000) in avoiding a turbulent bottom layer in the trap.

The traps were made out of 9.5-cm diameter Plexiglas tubing. Each tube was 51 cm high. Holes were drilled at appropriate levels on each tube to allow for draw-off of water after retrieval. As well, one hole was drilled in the bottom to allow for sediment removal. All holes were plugged with rubber stoppers during deployment.

There were eight traps used at each site. Three traps were used for collection of sedimenting material (sediment traps), another three contained random samples of zebra mussels for the purpose of collecting biodeposits (biodeposition traps), and two other traps contained empty shells of zebra mussels which were the controls for the biodeposition traps (control traps). Mussels in the biodeposition traps were all collected from the bottom at buoy E10 (Fig. 2.1). Each trap contained a similar number of mussels (300-500). The mussels were placed in a Plexiglas cage (Fig. 2.2) which fitted closely on the inside of the sediment trap. The cage was held in the trap approximately 10 cm from the top by a top and bottom peg which ran through the tube. Prior to deployment, all traps and cages were thoroughly cleaned with soap and water, and hexane-rinsed.

High resuspension will occur throughout the western basin of Lake Erie. Therefore, despite the use of open water sites in the present study, resuspension will still be a major contributing factor to the trap material. However the trap arrangement described previously enables one to compare the three types of traps with each other, since the effects of resuspension will be constant for each.

The trap deployment structure at each site consisted of a large Styrofoam buoy (473 mm x 321 mm) submerged 3 m below the surface of the water and anchored by approximately 200 kg of weight (Fig. 2.2). Galvanized aircraft cable (3 mm diam.) was used as a line to which the sediment traps were attached, onto fixed brackets, and held in place with elastic cords. All attachments (at the buoy, anchor, and sediment trap brackets) were constructed so that complete 360 degree movement around the line was possible. The approximate upward lift created by the buoy was 50 kg, which caused minimal swaying motion of the apparatus. This design proved to be effective against waves and water movement caused by internal currents or storm events.

Trap deployment took place August 11, 12, and 18, 1992, and they were left in the water until September 12, 1992. Traps were not disturbed during this time.

During the trap deployment, the trap material was not preserved as it settled into the bottom of the traps. In other studies, chloroform has been used to prevent any decomposition of trapped material (Baker et al. 1991, Broman et al. 1989). Sigg et al. (1987) suggested that using chemical preservatives in traps may interfere with contaminant levels in the settled material. This may be particularly true with organic contaminants, since chloroform, for example, is an organic compound which may sorb contaminants from the sediments or water. Therefore, for the present study no preservatives were used, since the potential effects of these on the results relating to contaminant values are unpredictable. As well, all rubber stoppers used in plugging drainage holes were first wrapped in hexane rinsed aluminum foil in order to prevent any contamination/sorption which may be caused by the rubber material.

2.2.2 Trap Removal and Sediment Sample Collection

After the deployment period (approximately 3-4 weeks), traps were removed from the water by SCUBA divers. While still attached to the line underwater, all traps were tightly capped to avoid any disturbance during ascent of the divers and during transport back to shore. Throughout retrieval and transport, material in traps did not appear to be disturbed. However, once on shore, all traps were left intact for a minimum of 5 h so that material which may have been suspended in the trap water would settle out. Very fine suspended material may not have settled out during this time, but it was assumed that most of the potential suspended material would have. In addition, the rate of settling for particles would be constant between trap types; therefore, comparisons between various trap data (described later in this methods section) would not be biased for any one trap type. Water was then drained off by removing successive stoppers from the top down along each tube. The small amount of remaining water and all trapped material was then removed by unplugging the bottom rubber stopper and letting the contents drain into a 1-L hexane-rinsed, acid-washed mason jar (see Appendix I for cleaning procedures). Any excess material left in the trap was rinsed through the bottom opening with water that was previously drained from the tube. All jars were appropriately labelled and capped with hexane-rinsed aluminum foil used as a liner. Upon return to the laboratory, all jars were placed in a refrigerator (4°C) for at least 12 h to allow any remaining suspended material to settle out. Water was then drained off by siphoning. The solid material was then thoroughly mixed with a stainless steel, hexane-rinsed, acid-washed probe. A small subsample (approx. 100 mg) was removed and immediately frozen (-22°C) for the purpose of determining percent organic matter content of the trap sample. This was done by drying each subsample at 60°C until constant weight (approx. 24 h), then weighed and ashed at 550°C for two hours after gradually bringing the samples up to temperature over a 3 h period. After being cooled down in a desiccator, samples were reweighed to determine the percent loss on ignition.

Blank determinations for potential Cd and PCB background contamination from the sampling equipment were not done in the study. Despite this, any potential contamination was assumed to be minimal since all equipment was appropriately cleaned. Also, if there was any contamination, this would be constant between samples since the same type of sampling procedures and material were used throughout the study; therefore, comparisons between data would not be affected.

Upon retrieval of the apparatus, mussels in the biodepositional traps were removed and immediately placed on ice until returned to the laboratory (same day) where they were frozen at -22°C. At a later date, the shell lengths of 150-200 mussels from each sample were determined using a Hipad™ digitizing pad and an associated computer program developed by Mr. Russel Hopcroft, University of Guelph. The shell lengths were used to determine the length-frequency distribution of mussels in each sample. The dry weight (referred to as DW from here on) of all mussel samples was then determined by drying at 60°C until constant weight was attained (approximately 48 h). The ash free dry weight (referred to as AFDW from here on) of each sample was determined by gradually bringing the samples up to 550°C over a 3 h period and then ashing at 550°C for 2 h. Samples were placed in a desiccator prior to re-weighing. These data were later used in calculating the amount of mussel biodeposits on a per unit weight basis.

2.2.3 Contaminant Analysis

All sediment samples were analyzed for total PCBs (t-PCBs) and Cd by Ortech Corporation, Mississauga, Ontario. For both t-PCB and Cd analysis, a separate portion of each sample was removed and accurately weighed. Moisture content was then determined. For t-PCB analysis, the sediment sample was triple extracted with 50% acetone in hexane. Cleanup procedures were performed on the extract to remove co-extracted material (e.g. lipids, oils, pigments) which would interfere with the quantification of the PCBs. This was done on a 2% water-deactivated Florisil column. Total PCB determinations were carried out by packed-column GC/EC technique. Aroclor 1254 was used as a PCB reference. For Cd analysis, the sample was digested twice with aqua regia ($3HCl:1\ HNO_3$), filtered through Whatman 40 paper, and then diluted to volume. Analysis was carried out by flame atomic absorption spectroscopy.

For quality assurance and control (QA/QC), two randomly chosen samples were split and submitted as regular samples. Results for contaminant analysis showed mostly similar values (Table 2.2). As part of Ortech's own QA/QC, several samples were analyzed twice, also showing very similar values (Table 2.3). As well, for PCB analysis a soil reference material (lot#9701) from Environmental Resource Associates was included, which gave 91% recovery.

2.2.4 Environmental Factors

Environmental factors were measured at each site during trap deployment, with sampling dates at the start and completion of the study, as well as once or twice between sample dates, depending on the site. The factors measured were total seston, organic seston, inorganic seston, chlorophyll a (chl a), temperature and dissolved oxygen. At each sampling date, two replicates for each factor were taken.

Seston samples were size fractionated into <53 μm and >53 μm. Ten Winkel and Davids (1982) reported that mussels preferentially select for particles between 15 and 40 um in size; therefore, size fractionation was done to better define the size range that mussels ingest. All samples were filtered through ashed, pre-weighed Whatman GF/C filters. Organic and inorganic content determinations of each of the size fractions was determined by ashing dried (60°C for 48 h), pre-weighed samples at 550°C for two hours after gradually bringing the samples up to this temperature over a 3 hour period. After ashing, samples were placed in a desiccator to allow for cooling prior to re-weighing. Chlorophyll a determinations were carried out using the fluorometric method (Parsons et al. 1984), and included correction for phaeopigments. Temperature and dissolved oxygen were determined using a YSI dissolved oxygen meter (Model 57), and a YSI probe (Model 5739).

2.2.5 Settlement Rack Study

At eight of the ten sites in the western basin (the exception being Point Pelee), several settlement racks were deployed in early June, 1992 for the purpose of determining growth rates and recruitment densities of mussels. These were deployed at the same depths as the sediment traps. Settlement rack construction is outlined in Fig. 2.3. The rack consisted of 5 Plexiglas plates (each plate 5.5 cm x 13.3 cm) mounted in a frame made of CPVC pipe (1.9 cm diameter). One rack was removed approximately every two weeks using SCUBA divers, thus enabling growth and recruitment density measurements to be determined for the entire rack deployment period (approximately two months). For each date, the number of mussels and the length/frequency distribution on each plate was determined. Length/frequency distributions were determined by digitizing 150-200 mussels per plate, or all mussels if numbers were less than this, using methods already described. From this, abundances of settled mussels and growth rates of mussels for each site could be determined.

2.2.6 Hamilton Harbour

All the above methods and materials, with the exception of the settlement rack study, were repeated for one location (two sites - top and bottom) in Hamilton Harbour, Ontario from September 29 to November 11, 1992. Weighted lines were suspended off the west pier at Canada Centre for Inland Waters (C.C.I.W.), Hamilton, Ontario, in 9.5 m of water. Sediment traps, arranged in the same manner as described for the western basin of Lake Erie, were attached 1 m below the surface (site 1, top) and 8.5 m below the surface (site 2, bottom). Water samples for measurement of environmental factors (see previous description) at each site were taken approximately every two days during this period. All contaminant analyses were carried out by Ortech Corporation by methods already described. In these analyses the PCB soil reference material was from the same lot number and gave 93% recovery.

2.2.7 Calculations and Statistical Analysis

An attempt was made to keep the number of mussels placed in the sediment traps at a density similar to or lower than those naturally found. This allowed for realistic comparisons between traps with and without mussels in order to determine if zebra mussels increase sedimentation rates.

Material produced by zebra mussels in the traps consisted of faeces and pseudofaeces, which were indistinguishable and therefore were included under the one term, biodeposits. Biodeposition refers to the process of the zebra mussels depositing suspended material from the water column. Sedimentation refers to the process of suspended material settling into the bottom of the sedimentation traps. Deposition refers to the process of suspended material settling into the bottom of traps through sedimentation and/or biodeposition. Biodeposited material was also indistinguishable from sedimented material in the traps containing zebra mussels. Therefore, biodeposition rates by mussels were determined for a site by subtracting the mean weight of material in the sedimentation and control traps (there was no significant difference ($p < 0.05$) over sites of these two types in weights and contaminant concentrations, and were combined for each site, appendices II, III, IV) from the weight of the material in each of the biodeposition traps. Each value was then divided by the ash-free dry weight of the mussels in the biodeposition trap and by the number of deployment days to determine the average daily biodeposition rate on a per unit tissue weight basis. Contaminant concentrations in the biodeposited material also could not be directly determined and were calculated using the following equation:

$$[\text{contaminant}]_{pf} = \frac{[\text{contam}]_{bd}*[\text{weight}]_{bd} - [\text{contam}]_{sm}*[\text{weight}]_{sm}}{\text{weight of pf}}$$

where: pf = pseudofaeces and faeces produced by mussels
sm = sedimented material (mean of sediment and control trap concentrations; see above)
bd = biodeposition trap material i.e. pf and sm in trap containing mussels
weight of pf = weight of bd - mean weight of sm

Biodeposition rates of contaminants per unit weight of mussel were also determined. This was done by multiplying the contaminant concentration of 'pf' by the biodeposition rate for each site. Sedimentation rates of contaminants were also calculated by multiplying the sedimentation rate (weight of sediment trap material divided by the number of deployment days) by the contaminant concentration of sm for each site.

The potential effect of zebra mussels on the deposition of suspended material and contaminants relative to sedimentation of these was assessed. To do this, deposition rate (i.e. the rate of deposition of all trap material in the biodeposition traps) and deposition rate of contaminants (deposition rate multiplied by the contaminant concentration of total trap material in the biodeposition trap) were compared to the respective sedimentation values using a paired Student t-test at $p \leq 0.05$ level of acceptance (Snedecor and Cochran 1989).

Biodeposition rates were compared with sedimentation rates to determine if there was any significant effect caused by zebra mussels. This comparison was done by converting biodeposition and sedimentation rates to a per unit area basis (i.e. $/m^2$).

Sedimentation rates in this respect were calculated by simple conversion from the area of the sediment trap opening. Biodeposition rates were converted to a per unit area value by first calculating the weight distribution of those mussels in the traps which were digitized using a length-weight regression equation (Mackie 1993). This equation is $y = 0.007x^{2.982}$ where 'x' is the shell length and 'y' is the dry tissue weight. For each biodeposition trap this weight was then multiplied by the number of mussels per m^2, determined from those found on the settlement racks on the last day of the sampling period (September 12, 1992), and then divided by the number of mussels digitized in the trap. This calculated weight per m^2 was multiplied by the biodeposition rate per unit weight of mussel tissue (calculation described previously) from the biodeposition traps for each site. If more settlement of mussels was to occur after this date, then at worst these calculated values would be underestimates of the tissue weight of mussels per unit area. Therefore, this was felt to be the most appropriate method of estimating biodeposition on a per unit area basis. Biodeposition per unit area was compared to sedimentation per unit area using a paired Student t-test at $p \leq 0.05$ level of acceptance.

Contaminant concentrations in the biodeposits and sediments were compared over all sites by using a paired Student t-test at the $p \leq 0.05$ level.

Biodeposition and sedimentation rates of contaminants were also compared on a per unit area basis by multiplying biodeposition and sedimentation rates per m^2 by the respective contaminant concentrations at each site. Comparisons were then made using a paired Student t-test at the $p \leq 0.05$ level.

Biodeposition rates of contaminants by mussels on a per unit weight basis were correlated with environmental factor data using multiple linear regression (SAS 1987). The level of acceptance was set at $p \leq 0.05$.

2.3 RESULTS

2.3.1 Biodeposition of Suspended Particulates

Biodeposition rates for Hamilton Harbour were very low, with rates being 0.02 ± 0.02 g DW biodeposits/ g AFDW tissue/d for the top site, and 0.01 ± 0.01 g DW biodeposits/g AFDW tissue/d for the bottom site. Overall rates of deposition of total trap material in the biodeposition and sedimentation traps (averaged for top and bottom sites) were 0.50 ± 0.10 g DW/d and 0.52 ± 0.07 g DW/d respectively.

Over sites in the western basin of Lake Erie, the tissue biomass of zebra mussels per m^2 was significantly greater (paired student t-test $p<0.05$) for the values at each site calculated from the settlement racks than for the densities of zebra mussels placed in the traps. Rates of deposition of the total material in the biodeposition and sediment traps for the western basin are presented in Table 2.4. When compared over all sites (paired Student t-test, $p<0.001$) the biodeposition traps had a significantly greater rate of deposition of material than that of the sediment traps.

The biodeposition rates of mussels for all sites in the western basin of Lake Erie are presented in Table 2.5. Biodeposition rates were weakly correlated ($p=0.071$) with suspended inorganic matter content <53 μm only ($R^2=0.351$) (Fig. 2.4, Appendix V). Other environmental factors showed no relationships ($p>0.10$) with biodeposition rates.

The biodeposition rate of mussels per unit area in the western basin is based on the calculated zebra mussel densities (from settlement racks) and biodeposition rate of mussels in the traps at each site. Zebra mussel densities for September 12, 1992, are shown for each site (except Point Pelee) in Table 2.6. When compared over sites (paired Student t-test, $p<0.05$), calculated biodeposition rates per unit area by zebra mussels were significantly greater than sedimentation rates per unit area (Table 2.7). Comparisons between sedimentation rates and biodeposition rates per unit area was not possible for Hamilton Harbour sites because no settlement racks were used.

2.3.2 Organic Matter Deposition

Organic matter data pertaining to Hamilton Harbour were not collected, therefore the following results in this respect will deal with the western basin of Lake Erie only.

Rates of deposition of organic matter in the biodeposition and sedimentation traps for the western basin of Lake Erie are presented in Table 2.8. When compared over all sites the biodeposition traps had a significantly greater ($p<0.001$) rate of deposition of organic matter than that of the sedimentation traps. There were no significant differences ($p>0.05$) between organic matter concentration in the biodeposited and sedimented material among western basin sites (Table 2.9). The calculated biodeposition rate of organic matter per unit area was significantly greater ($p<0.05$) than sedimentation rate of organic matter for sites in the western basin (Table 2.10).

2.3.3 PCB Biodeposition

Rates of deposition of PCBs in the biodeposition and sediment traps for the western basin of Lake Erie are presented in Table 2.11. When compared over all sites (paired

Student t-test, $p<0.001$), the biodeposition traps had a significantly greater rate of deposition of PCBs than that of the sediment traps.

There was no significant difference (paired Student t-test, $p>0.05$) between PCB concentration in the biodeposited and sedimented material over the sites in the western basin of Lake Erie. However, there appeared to be a trend of higher PCB levels in biodeposited material (Table 2.12). For Hamilton Harbour the biodeposits appeared to have higher PCB concentrations than the sedimented material although this was not statistically evident, possibly due to a low n (2) value (Table 2.12).

The calculated biodeposition rate of PCBs per unit area was significantly greater (paired Student t-test, $p<0.05$) than the sedimentation rate of this contaminant per unit area over sites in the western basin (Table 2.13). Comparison between sedimentation and biodeposition of PCBs per unit area was not possible for Hamilton Harbour because no settlement racks were used.

2.3.4 PCB Biodeposition Model

Biodeposition rates of PCBs per unit weight of zebra mussel were calculated from biodeposition trap data for the western basin of Lake Erie and for Hamilton Harbour (Table 2.14). PCB biodeposition rates for the western basin were regressed against environmental factors (Appendix VI) through multiple regression analysis (SAS 1987). The best fitting model for PCB biodeposition rates incorporated the factors organic matter content <53 μm (OM), inorganic matter content <53 μm (IM), and their interaction (IM*OM) ($p=0.0416$) (Appendix VII). The equation is:

$$y = 0.176 - (0.046)IM - (0.171)OM + (0.053)IM*OM \qquad R^2=0.722$$

Normality tests (stem-leaf plots, box plots) showed data to be normally distributed.

A graphical representation of this model is presented in Figure 2.5. High biodeposition rates of PCBs occurred at the highest levels of organic matter and inorganic matter (point A, Fig. 2.5), as well as at the lowest levels of both these factors (point B, Fig. 2.5). As the level of either factor decreased (from point A), or increased (from point B), the biodeposition rates of PCBs decreased, producing a saddle-shaped surface plot (Fig. 2.5).

2.3.5 Cd Biodeposition

Rates of deposition of Cd in the biodeposition and sediment traps for the western basin of Lake Erie are presented in Table 2.15. When compared over all sites (paired student t-test, $p<0.001$), the biodeposition traps had a significantly greater rate of deposition of Cd than that of the sedimentation traps.

There was no significant difference (paired Student t-test, $p<0.05$) between Cd concentration in the biodeposited and sedimented material over the sites in the western basin (Table 2.16). For Hamilton Harbour the biodeposits appeared to have higher contaminant concentrations than the sedimented material, although this was not statistically evident possibly due to a low n (2) value (Table 2.16).

The calculated biodeposition rate of Cd per unit area was significantly greater (paired Student t-test, $p<0.001$) than the sedimentation rate of this contaminant per unit area over sites in the western basin of Lake Erie (Table 2.17). Comparison between sedimentation and biodeposition rates of Cd per unit area was not possible for Hamilton Harbour because no settlement rack data were available.

2.3.6 Cadmium Biodeposition Model

Biodeposition rates of Cd per unit weight of zebra mussel were calculated from biodeposition trap data for the western basin of Lake Erie and Hamilton Harbour (Table 2.18). Cadmium biodeposition rates by zebra mussels in the western basin were regressed against environmental factors (Appendix VI) through multiple regression analysis (SAS 1987). Cadmium biodeposition was best related to inorganic matter <53 um, chl a (CHL), and the interaction (IM*CHL) ($p=0.0019$) (appendix VIII). The equation is:

$$y = 0.844 - (0.144)IM - (0.319)CHL + (0.089)IM*CHL \quad R^2=0.903$$

Normality tests (stem-leaf plots, box plots) showed data to be normally distributed.

A Graphical representation of this model is presented in Fig. 2.6. Cadmium deposition showed a similar trend to the PCB deposition model (Fig. 2.5). High biodeposition rates of Cd occurred at the highest levels of chl a and inorganic matter (point A, Fig. 2.6), as well as at the lowest levels of both these factors (point B, Fig. 2.6). As the level of either factor decreased (from point A) or increased (from point B) the biodeposition rates of Cd decreased, producing a saddle shaped surface plot (Fig. 2.6).

2.4 DISCUSSION

2.4.1 Biodeposition versus Sedimentation

Zebra mussels appear to increase the sedimentation of suspended material over natural sedimentation through their filtration and biodeposition processes. It is important to consider the density of mussels when comparing deposition rates with sedimentation rates. Within the sediment traps mussel densities were significantly lower than the estimated population densities for each site calculated from the settlement racks. Therefore the deposition rate in the biodeposition traps was greater than that found in the sediment traps even at the mussel densities used in the biodeposition traps.

Other studies have shown different results when comparing biodeposition by mussels to natural sedimentation, the differences appearing to be due to season. Haven and Morales-Alamo (1966), in a controlled field experiment, observed greater amounts of deposited material in trays holding oysters (Crassostrea virginica) compared to control trays with no oysters. Kautsky and Evans (1987), using a similar experimental design to the present study, saw greater deposition in traps containing mussels compared to control traps without mussels, for the summer months; lower deposition rates observed from January to April were attributed to lower temperatures and thus lower metabolic activity by mussels during this time. Jaramillo et al. (1992), also using sediment traps, observed lower or similar biodeposition rates to sedimentation rates, also depending on the time of year. They suggest that this was also due to decreased water temperatures and to low water salinities. In these latter two studies experimental mussel densities were representative of those found in the surrounding study area. In the present study, experiments were conducted during the summer and fall. During the summer study, deposition in traps containing mussels was much greater than sedimentation in the sediment traps (average deposition rate of 1.92 ± 0.12 g DW/d versus average sedimentation rate of 0.82 ± 0.21 g DW/d), but these were similar during the fall study in Hamilton Harbour (average deposition rate of 0.50 ± 0.10 g DW/d versus average sedimentation rate of 0.52 ± 0.07 g DW/d). Interestingly, Kautsky and Evans (1987) observed the highest biodeposition rates during the fall, suggesting that the increased turbidity at this time caused an increase in biodeposition rate. Deslous-Paoli et al. (1992) suggested that increased turbidity may have a 'shutting down' effect on the mussel, thus lowering biodeposition rate. This is perhaps what has happened in the present study because high turbidity was observed during this period. The effects of time of year may play an important role on zebra mussel deposition, perhaps due to temperature and/or suspended particulate concentration differences.

To quantify the effects of biodeposition on a lake, biodeposition rates must be converted to a per unit area basis so that direct comparisons can be made with sedimentation rates. Tsuchiya (1980) calculated this for Mytilus edulis in intertidal and subintertidal regions of Japan, by developing regression models of mussel shell length versus mussel biodeposition for different times of the year, then applying this to *in situ* mussels. In the present study a different approach was taken where biodeposition rates determined from trap mussels were multiplied by the tissue weight of mussels on a per m^2 basis given the density found on the settlement racks. This was felt to be the most appropriate approach here since it takes into account the biodeposition rate of a mature population of mussels (mussels used in the biodeposition traps) and the density of the mussels found at the site of interest (newly settled mussels on the racks). This method may actually be overestimating the true biodeposition, since the density values of mussels are based on the settlement rack data, which may not be representative of natural densities. For example, racks suspended in the water column may have less predation exposure than do *in situ* mussels on the bottom of the lake, resulting in a higher density estimate than that naturally found. However density measurements of *in situ* mussels were not taken in this study, therefore the settlement rack data are the best available estimates for determining specific site densities. Leach (1993) recorded densities of *in situ* zebra mussels at between 142,000 to 342,000/m^2 on reefs in western Lake Erie. The densities estimated from the settlement racks in the present study (see Table 2.6) are similar to Leach's values; therefore, results incorporating settlement rack data are applicable to western basin locations with solid substrates, such as reefs.

No other studies to this date have attempted to relate the effects of biodeposition rates by zebra mussels to sedimentation rates on a per unit area basis. When this is considered, the effects of mussels appear quite pronounced in comparison to natural sedimentation (see results, chapter two). Several studies have outlined the potential effects of zebra mussels on water clarification. Reeders et al. (1989) concluded that their experimental lakes could be filtered at least once or twice a month by the existing zebra mussel population. In Lake Erie, Holland (1993) observed lower plankton levels in water samples taken after the zebra mussel invasion compared with pre-invasion data. MacIsaac et al. (1992) also observed a depletion of chl a concentrations above Dreissena beds, attributing this to their filtration. These findings, in conjunction with the present results, suggest that filtration and resulting biodeposition by zebra mussels in Lake Erie is potentially great enough to alter water column biomass of suspended material.

Biodeposition by zebra mussels was positively correlated with suspended inorganic matter content. In general, this is in agreement with other studies. Deslous-Paoli et al. (1992) observed a linear correlation between pseudofaecal production and total suspended material for the mollusc Crassostrea gigas. Similar results were observed by Foster-Smith (1975), Tenore and Dunstan (1973) and Haven and Morales-Alamo (1966) for various bivalve species. Pseudofecal production has been attributed in part to the presence of inorganic matter (Sprung and Rose 1988, Kautsky and Evans 1987), which is selected against and discarded during filtration. In the present study the relationship between inorganic matter and total biodeposits may suggest that mussel biodeposits contain a large proportion of psuedofeces. This is in agreement with Reeders and Bij de Vaate (1992) who observed that 90% of the biodeposits by zebra mussels in their study was pseudofeces.

2.4.2 Organic matter, PCB and Cd deposition

The results of this study support the first prediction of objective one that zebra mussels significantly increase the deposition rate of PCBs and Cd over that of natural sedimentation. Results pertaining to PCB and Cd deposition by zebra mussels followed a similar trend, and therefore will be included together in this discussion.

The rate of organic matter deposition was significantly greater than natural sedimentation rates of organic matter in the trap comparisons. This has particular importance, among other reasons, because increased deposition of organic matter due to zebra mussels may be providing other benthic invertebrates with an increased food source (Kautsky and Evans 1987, Deslous-Paoli et al. 1992, Tsuchiya 1980). Kautsky and Evans (1987) point out that species diversity and biomass are generally higher in mussel beds than in areas without mussels. The same study, found that M. edulis increased deposition of organic matter to the bottom sediments. Izvekova and Lvova-Katchanova (1972) observed greater growth of chironomid larvae exposed to zebra mussel biodeposits than when exposed to other food sources. Therefore it appears that zebra mussels are depositing significant amounts of organic material through their biodeposition, which could then be providing an additional food source for benthic animals.

Organic matter concentration in the biodeposits was not significantly different from that of the sedimented material. Jaramillo et al. (1992) have found that the biodeposits from mussels used in their study were, in general, higher in organic carbon content than the sedimented material. Kautsky and Evans (1987) found similar results. The latter study attributed this to the selection by mussels of particles that had high organic content. It has also been suggested that pseudofeces production by mussels is a response to overall high concentrations of suspended material which may clog the gills (Foster-Smith 1975, Sprung and Rose 1988, Reeders and Bij de Vaate 1992). Although total suspended solid concentrations in the present study were not exceedingly high (maximum of 6.4 mg/l in the present study compared to a maximum of 86.4 mg/l in Reeders and Bij de Vaate 1992), perhaps zebra mussels are producing pseudofeces in response to these concentrations of suspended solids. The sedimentation of suspended material will include organic and inorganic components. Therefore biodeposits would be expected to be similar in organic matter concentrations to sedimented material, since here there would not be a specific selection for organic or inorganic particulates by mussels to be deposited as pseudofeces.

The concentrations of PCBs and Cd in the biodeposits were similar to those in the sediments. Reeders et al. (1993) observed a higher level of both of these contaminants in zebra mussel biodeposits than with suspended material. They attribute this to a selection for deposition of smaller particles, which would therefore have a greater surface area and greater contaminant concentration. The measurement of the size range of particles was not possible in the present study since naturally sedimenting material and biodeposits were mixed together in the traps. Both Cd and PCBs have been shown to preferentially bind to organic matter, especially PCBs which are by nature lipophilic (Servos and Muir 1989, Morfett et al. 1988, Allan 1986). As discussed, organic matter levels in the biodeposits and sediments were found to be quite similar, so contaminant concentrations may not differ because of the similarity in the organic matter content to which these contaminants are bound.

Given that the biodeposits have a similar contaminant concentration to the sedimented material, and biodeposition by mussels is significantly greater than sedimentation, it would be expected that the biodeposition of contaminants would exceed sedimentation of contaminants. Studies have examined the nutrient content of biodeposits of mussels (Deslous-Paoli et al. 1992), Jaramillo et al. 1992, Kautsky and Evans 1987) and effects of mussels on nutrient movement in lakes (Stanczykowska and Lewandowski 1993), but few studies have involved similar types of research with contaminants. Reeders and Bij de Vaate (1992) have not compared the biodeposition of contaminants with sedimentation, although they suggest that zebra mussels may act as an excellent 'biological filter' through the biodeposition of contaminants. The results of the present study support the prediction that mussels are increasing the sedimentation rate of PCBs and Cd in Lake Erie.

When the biodeposition rates of contaminants are converted to a per m^2 basis, the potential effects on the benthic food web become evident. It has already been suggested that increased deposition may be increasing food supply to benthic feeders. In so doing, contaminant transfer from pelagic areas to benthic food webs may be increasing. Alternatively, much of this material biodeposited out by zebra mussels could be resuspended, thus not being as available to benthic animals. Kautsky and Evans (1987) stated that mussel biodeposits consist of packages of fine sized particles which can easily be broken up and resuspended. This may be particularly true of pseudofecal pellets since these are not as compacted as fecal pellets, and the mucus exterior can easily dissolve in water, thus allowing for easy breakup and resuspension (Deslous-Paoli et al. 1992). Kemp et al. (1977) observed a mean sedimentation rate for western Lake Erie of 5.92 g/m^2/d, which is considerably less than the sedimentation rate observed in the present study (see Table 2.7). Therefore this suggests that much of the material being trapped is from resuspension. If most of the biodeposition by zebra mussels consists of pseudofeces, as suggested earlier, then the net effect of biodeposition may not be as great as shown in the present study, since the resuspension factor was not considered. Further studies dealing with the effects of contaminant biodeposition on benthic animals and food web transfer are required to understand these processes. These should include the use of experimental designs which attempt to differentiate between net sedimentation and resuspension of particulates, and thus the associated contaminants.

2.4.3 PCB Biodeposition Model

Results for the PCB model were partially in agreement with the prediction that PCB deposition by mussels will be positively correlated with total suspended solids concentration, food quantity and inorganic matter. Because of the nature of the model developed in this study, PCB biodeposition rate by zebra mussels was both negatively and positively correlated to organic and inorganic factors, depending on the relative concentrations of these.

Organic matter and inorganic matter content in the suspended material appear to be important variables in the model of biodeposition of PCBs. Results indicate that mussels are altering their filtration/biodeposition rates relative to the concentrations of inorganic matter and organic matter. High biodeposition rates of PCBs occur when the levels of organic and inorganic matter are highest, and also when they are lowest. Numerous studies have observed high biodeposition rates in mussels when the concentration of suspended material was high (Reeders and Bij de Vaate 1992, Haven and Morales-Alamo 1966, Deslous-Paoli et al. 1992). In the present study, the high PCB biodeposition rate at high levels of organic and inorganic material may be due to the high suspended particulate load, where mussels respond by biodepositing the contaminated suspended material. The overall concentration of suspended material will be low at low levels of inorganic and organic material. Foster-Smith (1975) observed higher filtration rates at low concentrations of suspended material than at high levels. Perhaps mussels are compensating for low food at lower concentrations by having a higher filtration rate, resulting in a high biodeposition rate.

Proportions of inorganic and organic matter may be important since changes in these appear to alter the biodeposition of PCBs. For instance, as organic matter levels increase in the model, while inorganic matter levels remain the same (low), there is a decrease in the biodeposition rate of PCBs, perhaps because more of the material, being organic, is assimilated as a food source. As inorganic matter levels increase while organic matter levels remain the same (low), biodeposition rate of PCBs decrease. Earlier results from the present study showed a positive correlation between inorganic matter and biodeposition rate by mussels. As mentioned, studies have shown that increasing inorganic matter levels increases biodeposition rate of pseudofeces (Deslous-Paoli et al. 1992, Foster-Smith 1975, Tenore and Dunstan 1973, Haven and Morales-Alamo 1966). Therefore, the lower biodeposition rates of PCBs with increasing inorganic matter is probably because PCBs have a low affinity for inorganic material. Hence, despite increased biodeposition rate by mussels, the amount of contaminant deposited decreased. Under such conditions the accumulation of PCBs by mussels should reflect this, that is, be low. This will also be discussed in later section.

2.4.4 Cadmium Biodeposition Model

The results from the Cd biodeposition model follow the same trends as those of PCBs. In the Cd model chl a was a more important factor than organic matter. High biodeposition rates of Cd occur at the highest levels of chl a and inorganic matter, as well as at the lowest levels of these. Biodeposition rates of Cd at these two combinations of factors are similar. If mussels have a high biodeposition rate at low levels of each factor in order to maintain a high level of food intake (see previous discussion) then Cd biodeposition also could be high because, as in the case with many metals, inorganic particulates such as manganese and iron oxides, are important in the sorption of these elements (Morfett et al. 1988, Sigg et al. 1987) in addition to organic particulates. Therefore the inorganic component of the biodeposits may also be contributing a certain amount of bound Cd. As chl a concentration increases in the model and inorganic matter concentration remains low, Cd biodeposition is low, which may be because mussels are ingesting the organic material (here represented as chl a). This should be reflected in the uptake of Cd by the mussels (i.e. high rate of Cd uptake), discussed later. As inorganic matter increases in the model and chl a remains low, biodeposition of Cd is also low. This is interesting because if Cd has a high affinity for certain components of inorganic material then an increased rate of biodeposition of suspended material (see discussion on PCB model) would result in an increased biodeposition rate of Cd. No definite explanation can be given for this, but it may be that the biodeposits are relatively low in organic content under these conditions (due to low levels of suspended organic material) and the biodeposition rate of Cd decreases because of a decreased contribution of organically bound Cd.

Table 2.1. Loran positions for all study locations in the western basin of Lake Erie.

location	Loran coordinates
E13	41, 51. 51' N 82, 51. 54' W
Pelee Island	41, 43. 36' N 82, 45. 93' W
Point Pelee	41, 59. 40' N 82, 34. 48' W
E10	41, 53. 12' N 82, 36. 47' W
buoy H	41, 52. 36' N 83, 04. 49' W

Table 2.2. PCB and Cd concentrations of settled trap material at sites Point Pelee bottom, and Pelee Island bottom, western basin of Lake Erie, August 1992. Each sample was split into two subsamples for quality assurance and control procedures. 'Bottom' refers to sites where sampling apparati were placed 1.5 m of the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	**PCB concentration (μg PCBs/g DW settled material)**	**Cd concentration (μg Cd/g DW settled material)**
Point Pelee biodeposition trap sample, bottom	0.08 0.23	0.80 0.74
Pelee Island biodeposition trap sample, bottom	0.10 0.09	0.74 0.63

Table 2.3. Duplicate measurements of PCB concentrations for settled trap material for various sites in the western basin of Lake Erie, August 1992, as part of quality assurance and control procedures. 'Top' refers to sites where sampling apparati were placed 3 m under the surface of the water, and 'bottom' refers to sites where sampling apparati were placed 1.5 m off the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	**PCB concentration (μg PCBs/g DW settled material)**
Point Pelee sediment trap sample, bottom, replicate #3	0.08 0.07
E10 biodeposition trap sample, top, replicate #2	0.25 0.20
E13 biodeposition trap sample, bottom, replicate #1	0.08 0.08
H biodeposition trap sample, bottom, replicate # 2	0.16 0.16

Table 2.4. Deposition rate of total trap material in biodeposition traps, and sedimentation rates of total trap material in sediment traps for all sites in the western basin of Lake Erie, August 1992. 'Top' refers to sites where sampling apparati were placed 3 m under the surface of the water, and 'bottom' refers to site where sampling apparati were placed 1.5 m off the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	**deposition rate in biodeposition traps (g DW/d) (mean ± 1 S.D.)**	**sedimentation rate in sediment traps (g DW/d) (mean ± 1 S.D.)**
E13 top	1.36 ± 0.20	0.79 ± 0.22
E13 bottom	1.38 ± 0.09	1.06 ± 0.30
Pelee Isand top	1.38 ± 0.30	0.59 ± 0.04
Pelee Isand bottom	2.44 ± 0.38	0.73 ± 0.06
Point Pelee top	2.52 ± 0.41	0.83 ± 0.67
Point Pelee bottom	3.61 ± 0.26	1.16 ± 0.12
E10 top	1.80 ± 0.03	0.75 ± 0.24
E10 bottom	2.24 ± 0.15	1.06 ± 0.12
H top	1.08 ± 0.13	0.60 ± 0.15
H bottom	1.42 ± 0.22	0.60 ± 0.09

Table 2.5. Biodeposition rates of suspended material by zebra mussels calculated from the biodeposition traps at all sites in the western basin of Lake Erie, August 1992. 'Top' refers to sites where sampling apparati are placed 3 m under the surface of the water and 'bottom' refers to sites where sampling apparati were placed 1.5 m off the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and coordinates.

site	**biodeposition rate (g DW biodeposits/g DW tissue/d) (mean ± 1 S.D)**
E13 top	0.096 ± 0.049
E13 bottom	0.055 ± 0.011
Pelee Island top	0.142 ± 0.098
Pelee Island bottom	0.446 ± 0.054
Point Pelee top	0.471 ± 0.112
Point Pelee bottom	0.799 ± 0.037
E10 top	0.296 ± 0.013
E10 bottom	0.383 ± 0.106
H top	0.089 ± 0.022
H bottom	0.179 ± 0.050

Table 2.6. Adult zebra mussel abundances (/m^2) calculated from mussels found on settlement racks at sites in the western basin of Lake Erie, September 12, 1992. 'Top' refers to sites where sampling apparati were placed 3 m under the surface of the water, and 'bottom' refers to sites sampling apparati were placed 1.5 m off the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	number/m^2
E13 top	1082263 ± 288367
E13 bottom	732208 ± 14151
Pelee Island top	291529 ± 115296
Pelee Island bottom	200456 ± 52834
E10 top	224072 ± 71566
E10 bottom	214010 ± 83128
H top	206775 ± 69887
H bottom	319264 ± 86515

Table 2.7. The calculated biodeposition rates of suspended material by zebra mussels on a per m^2 basis (refer to methods, chapter two, for calculation procedure), and sedimentation rates of suspended material on a per m^2 basis at all sites in the western basin of Lake Erie (August 1992) and Hamilton Harbour (September - November 1992). Unless otherwise stated, 'top' refers to sites where sampling apparati were placed 3 m under the surface of the water, and 'bottom' refers to sites where sampling apparati were placed 1.5 m off the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	**Biodeposition Rate (g DW biodeposits/m^2/d) (mean ± 1 S.D.)**	**Sedimentation Rate (g DW sediments/m^2/d) (mean ± 1 S.D.)**
E13 top	2076 ± 1062	90.3 ± 6.8
E13 bottom	613 ± 119	165.1 ± 42.9
Pelee Island top	636 ± 440	83.2 ± 7.1
Pelee Island bottom	1379 ± 169	105.8 ± 9.3
Point Pelee top	------	121.9 ± 7.1
Point Pelee bottom	------	167.9 ± 17.5
E10 top	774 ± 33	88.5 ± 5.6
E10 bottom	896 ± 248	150.1 ± 0.6
H top	236 ± 58	77.9 ± 19.5
H bottom	645 ± 181	91.7 ± 14.7
Hamilton Harbour top (1 m below the surface)	------	36.4 ± 0.71
Hamilton Harbour bottom (8.5 m below the surface)	------	103.3 ± 26.5

Table 2.8. Deposition rates of total organic matter (OM) in biodeposition traps, and sedimentation rates of total organic matter in sediment traps for all sites in the western basin of Lake Erie, August 1992. 'Top' refers to sites where sampling apparati were placed 3 m under the surface of the water, and 'bottom' refers to sites where sampling apparati were placed 1.5 m off the bottom of Lake Erie. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	**deposition rate of organic matter in the biodeposition traps (g AFDW OM/d) (mean ± 1 S.D.)**	**sedimentation rate of organic matter in the sediment traps (g AFDW OM/d) (mean ± 1 S.D.)**
E13 top	0.125 ± 0.022	0.062 ± 0.018
E13 bottom	0.107 ± 0.065	0.092 ± 0.031
Pelee Island top	0.135 ± 0.034	0.044 ± 0.011
Pelee Island bottom	0.166 ± 0.112	0.061 ± 0.017
Point Pelee top	0.226 ± 0.055	0.074 ± 0.007
Point Pelee bottom	0.273 ± 0.014	0.094 ± 0.018
E10 top	0.141 ± 0.029	0.072 ± 0.024
E10 bottom	0.189 ± 0.015	0.092 ± 0.028
H top	0.119 ± 0.015	0.070 ± 0.023
H bottom	0.167 ± 0.028	0.065 ± 0.009

Table 2.9. Calculated organic matter (OM) concentrations in zebra mussel biodeposits from the biodeposition traps (refer to methods, chapter two, for equation), and in settled material in sediment traps at all sites in the western basin of Lake Erie, August 1992. 'Top' refers to sites where sampling apparati were placed 3 m under the surface of the water, 'bottom' refers to sites where sampling apparati were placed 1.5 m off the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	organic matter concentration in biodeposits (g OM/g DW biodeposits) (mean ± 1 S.D.)	organic matter concentration in sediment (g OM/g DW sediment) (mean ± 1 S.D.)
E13 top	0.110 ± 0.009	0.079 ± 0.006
E13 bottom	0.163 ± 0.015	0.085 ± 0.009
Pelee Island top	0.115 ± 0.015	0.075 ± 0.015
Pelee Island bottom	0.097 ± 0.008	0.076 ± 0.004
Point Pelee top	0.099 ± 0.011	0.090 ± 0.006
Point Pelee bottom	0.074 ± 0.004	0.073 ± 0.005
E10 top	0.067 ± 0.024	0.092 ± 0.009
E10 bottom	0.088 ± 0.001	0.085 ± 0.018
H top	0.101 ± 0.018	0.115 ± 0.011
H bottom	0.118 ± 0.009	0.110 ± 0.010

Table 2.10. The calculated biodeposition rates of organic matter (OM) on a per m^2 basis, and sedimentation rates of organic matter on a per m^2 basis at all sites in the western basin of Lake Erie, August 1992. 'Top' refers to sites where sampling apparati were placed 3 m under the surface of the water, and 'bottom' refers to sites where sampling apparati were placed 1.5 m off the bottom of Lake Erie. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	**biodeposition rate of organic matter (g AFDW OM/m^2/d) (mean ± 1 S.D.)**	**sedimentation rate of organic matter (g AFDW OM/m$^{2/}$d) (mean ± 1 S.D.)**
E13 top	224.2 ± 111.4	7.0 ± 1.1
E13 bottom	101.3 ± 31.8	13.9 ± 4.8
Pelee Island top	73.0 ± 50.8	7.2 ± 0.8
Pelee Island bottom	133.2 ± 26.2	9.6 ± 2.8
E10 top	51.5 ± 17.4	8.0 ± 3.6
E10 bottom	79.6 ± 23.1	11.8 ± 0.6
H top	23.7 ± 7.0	8.7 ± 2.2
H bottom	88.3 ± 38.2	9.3 ± 1.5

Table 2.11. Deposition rates of PCBs in biodeposition traps, and sedimentation rates of PCBs in sediment traps for all sites in the western basin of Lake Erie, August 1992. 'Top' refers to sites where sampling apparati were placed 3 m under the surface of the water, and 'bottom' refers to sites where sampling apparati were placed 1.5 m off the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	deposition rate of PCBs in the biodeposition traps (μg PCBs/d) (mean ± 1 S.D.)	sedimentation rate of PCBs in the sediment traps (ug PCBs/d) (mean ± 1 S.D.)
E13 top	0.22 ± 0.04	0.09 ± 0.05
E13 bottom	0.12 ± 0.03	0.17 ± 0.06
Pelee Island top	0.34 ± 0.09	0.14 ± 0.08
Pelee Island bottom	0.20 ± 0.01	0.16 ± 0.08
Point Pelee top	0.33 ± 0.12	0.08 ± 0.01
Point Pelee bottom	0.24 ± 0.01	0.10 ± 0.02
E10 top	0.36 ± 0.09	0.11 ± 0.03
E10 bottom	0.38 ± 0.12	0.16 ± 0.03
H top	0.17 ± 0.03	0.11 ± 0.04
H bottom	0.24 ± 0.03	0.08 ± 0.03

Table 2.12. Calculated PCB concentrations in the zebra mussel biodeposits from biodeposition traps (refer to methods, chapter two, for calculation procedures), and in settled material in sediment traps at all sites in the western basin of Lake Erie (August 1992), and Hamilton Harbour (September - November 1992). Unless otherwise stated, 'top' refers to sites where sampling apparati were placed 3 m under the surface of the water, and 'bottom' refers to sites where sampling apparati were placed 1.5 m off the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	**PCB concentration in the biodeposits (μg PCBs/g DW biodeposits) mean ± 1 S.D.)**	**PCB concentration in the sediment (μg PCBs/g DW sediment) (mean ± 1 S.D.)**
E13 top	0.240 ± 0.072	0.114 ± 0.025
E13 bottom	0.090 ± 0.028	0.154 ± 0.038
Pelee Island top	0.276 ± 0.058	0.202 ± 0.121
Pelee Island bottom	0.037 ± 0.015	0.220 ± 0.112
Point Pelee top	0.163 ± 0.122	0.098 ± 0.015
Point Pelee bottom	0.104 ± 0.071	0.083 ± 0.010
E10 top	0.258 ± 0.076	0.122 ± 0.029
E10 bottom	0.212 ± 0.076	0.122 ± 0.019
H top	0.102 ± 0.058	0.192 ± 0.055
H bottom	0.177 ± 0.023	0.154 ± 0.026
Hamilton Harbour top (1 m below the surface)	0.557 ± 0.040	0.444 ± 0.026
Hamilton Harbour bottom (8.5 m below the surface)	0.800 ± 0.298	0.328 ± 0.056

Table 2.13. The calculated biodeposition rates of PCBs per m^2 by zebra mussels (refer to methods, chapter two, for calculations), and the sedimentation rates of PCBs per m^2 for sites at sites in the western basin of Lake Erie (August 1992) and Hamilton Harbour (September - November 1992). Unless otherwise stated, 'top' refers to sites where sampling apparati were placed 3 m under the surface of the water, and 'bottom' refers to sites where sampling apparati were placed 1.5 m off the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	**Biodeposition rate PCBs (μg PCBs/m^2/d) (mean ± 1 S.D.)**	**Sedimentation rate PCBs (μg PCBs/m^2/d) (mean ± 1 S.D.)**
E13 top	480.3 ± 228.1	8.8 ± 1.0
E13 bottom	50.7 ± 8.0	26.8 ± 4.7
Pelee Island top	177.3 ± 123.4	13.4 ± 8.8
Pelee Island bottom	49.5 ± 13.1	20.5 ± 12.1
Point Pelee top	------	12.6 ± 1.1
Point Pelee bottom	------	13.3 ± 2.8
E10 top	197.9 ± 51.9	9.6 ± 0.9
E10 bottom	198.8 ± 119.1	16.7 ± 1.3
H top	25.2 ± 15.6	13.8 ± 1.3
H bottom	117.7 ± 21.9	15.4 ± 1.3
Hamilton Harbour top (1 m below the surface)	------	16.5 ± 0.8
Hamilton Harbour bottom (8.5 m below the surface)	------	37.8 ± 11.4

Table 2.14. Biodeposition rates of PCBs per unit weight of dry zebra mussel tissue calculated from biodeposition traps at all sites in the western basin of Lake Erie (August 1992), and Hamilton Harbour (September - November 1992). Unless otherwise stated, 'top' refers to sites where sampling apparati were placed 3 m under the surface of the water, and 'bottom' refers to sites where sampling apparati were placed 1.5 m off the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	**biodeposition rates of PCBs (μg PCBs biodeposited/g DW tissue/d (mean ± 1 S.D.)**
E13 top	0.022 ± 0.011
E13 bottom	0.005 ± 0.001
Pelee Island top	0.040 ± 0.003
Pelee Island bottom	0.016 ± 0.004
Point Pelee top	0.068 ± 0.033
Point Pelee bottom	0.085 ± 0.061
E10 top	0.076 ± 0.020
E10 bottom	0.085 ± 0.051
H top	0.010 ± 0.006
H bottom	0.031 ± 0.006
Hamilton Harbour top (1 m below the surface)	0.010 ± 0.001
Hamilton Harbour bottom (8.5 m below the surface)	0.010 ± 0.001

Table 2.15. Deposition rates of Cd in biodeposition traps, and sedimentation rates of Cd in the sediment traps for all sites in the western basin of Lake Erie, August 1992. 'Top' refers to sites where sampling apparati were placed 3 m under the surface of the water, and 'bottom' refers to sites where sampling apparati were placed 1.5 above the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	**deposition rate of Cd in biodeposition traps (ug Cd/d) (mean ± 1 S.D.)**	**Sedimentation rate of Cd in sediment traps (ug Cd/d) (mean ± 1 S.D.)**
E13 top	0.77 ± 0.21	0.67 ± 0.21
E13 bottom	0.69 ± 0.12	1.01 ± 0.35
Pelee Is. top	1.18 ± 0.33	0.28 ± 0.08
Pelee Is. bottom	1.63 ± 0.15	0.53 ± 0.16
Point Pelee top	3.28 ± 1.26	0.69 ± 0.10
Point Pelee bottom	2.99 ± 0.06	0.96 ± 0.14
E10 top	1.60 ± 0.19	0.60 ± 0.07
E10 bottom	2.29 ± 0.69	0.82 ± 0.17
H top	1.03 ± 0.09	0.48 ± 0.27
H bottom	1.44 ± 0.25	0.34 ± 0.15

Table 2.16. Calculated Cd concentrations in zebra mussel biodeposits from biodeposition traps (refer to methods, chapter two, for calculation procedure), and in settled material in sediment traps at all sites in the western basin of Lake Erie (August 1992), and Hamilton Harbour (September - November 1992). Unless otherwise stated, 'top' refers to sampling sites where sampling apparati are placed 3 m under the surface of the water, and 'bottom' refers to sampling apparati placed 1.5 m off the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	**Cd concentration in the biodeposits (μg Cd/g DW biodeposits) (mean ± 1 SD)**	**Cd concentration in the sediment (μg Cd/g DW sediment) (mean ± 1 SD)**
E13 top	0.35 ± 0.18	0.85 ± 0.10
E13 bottom	0.51 ± 0.12	0.96 ± 0.20
Pelee Island top	1.08 ± 0.19	0.48 ± 0.14
Pelee Island bottom	0.69 ± 0.05	0.72 ± 0.22
Point Pelee top	1.52 ± 0.67	0.84 ± 0.09
Point Pelee bottom	0.80 ± 0.07	0.83 ± 0.08
E10 top	0.83 ± 0.17	0.85 ± 0.28
E10 bottom	1.31 ± 0.75	0.77 ± 0.13
H top	0.75 ± 0.02	0.87 ± 0.41
H bottom	1.41 ± 0.36	0.61 ± 0.22
Hamilton Harbour top (1 m below the surface)	4.49 ± 2.36	3.00 ± 0.19
Hamilton Harbour bottom (8.5 m below the surface)	8.92 ± 3.38	2.50 ± 0.21

Table 2.17. The calculated biodeposition rates of Cd per m^2 by zebra mussels (refer to methods, chapter two, for calculation procedures), and sedimentation rates of Cd per m^2 for sites in the western basin of Lake Erie (August 1992), and Hamilton Harbour (September - November 1992). Unless otherwise stated, 'top' refers to sites where sampling apparati were placed 3 m under the surface of the water, and 'bottom' refers to sites where sampling apparati were placed 1.5 m above the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	**Biodeposition rate Cd (μg Cd/m^2/d) (mean ± 1 S.D.)**	**Sedimentation rate Cd (μg Cd/m^2/d) (mean ± 1 S.D.)**
E13 top	609.8 ± 145.8	79.4 ± 18.5
E13 bottom	303.9 ± 15.9	82.8 ± 27.3
Pelee Island top	689.9 ± 471.1	45.6 ± 7.9
Pelee Island bottom	944.3 ± 43.8	65.3 ± 28.0
Point Pelee top	------	107.4 ± 4.2
Point Pelee bottom	------	146.7 ± 9.9
E10 top	637.4 ± 105.4	80.7 ± 8.1
E10 bottom	1083.0 ± 344.0	154.9 ± 36.7
H top	177.6 ± 41.8	88.7 ± 41.7
H bottom	885.1 ± 191.7	55.6 ± 9.3
Hamilton Harbour top (1 m below the surface)	------	114.0 ± 1.1
Hamilton Harbour bottom (8.5 m below the surface)	------	267.6 ± 67.7

Table 2.18. Biodeposition rates of Cd per unit weight of dry zebra mussel tissue calculated from biodeposition traps at all sites in the western basin of Lake Erie (August 1992), and Hamilton Harbour (September - November 1992). Unless otherwise stated, 'top' refers to sites where sampling apparati were placed 3 m under the surface of the water, and 'bottom' refers to sites where sampling apparati were placed 1.5 off the bottom of the Lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	**biodeposition rates of Cd (μg Cd/g DW tissue/d (mean ± 1 S.D.)**
E13 top	0.028 ± 0.007
E13 bottom	0.027 ± 0.001
Pelee Island top	0.154 ± 0.105
Pelee Island bottom	0.445 ± 0.054
Point Pelee top	0.710 ± 0.371
Point Pelee bottom	0.641 ± 0.032
E10 top	0.244 ± 0.040
E10 bottom	0.463 ± 0.147
H top	0.067 ± 0.016
H bottom	0.246 ± 0.053
Hamilton Harbour top (1 m below the surface)	0.090 ± 0.015
Hamilton Harbour bottom (8.5 m below the surface)	0.104 ± 0.078

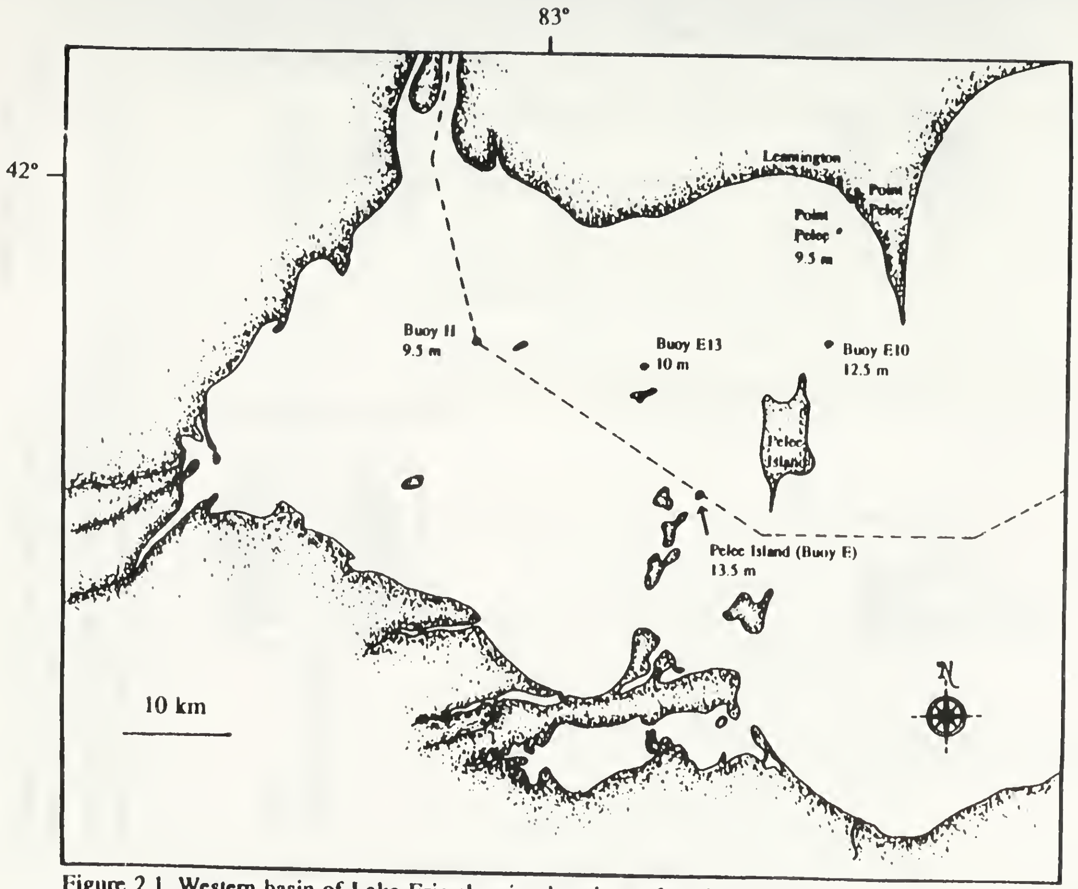


Figure 2.1. Western basin of Lake Erie showing locations of study, coast guard buoy names, and maximum depths.

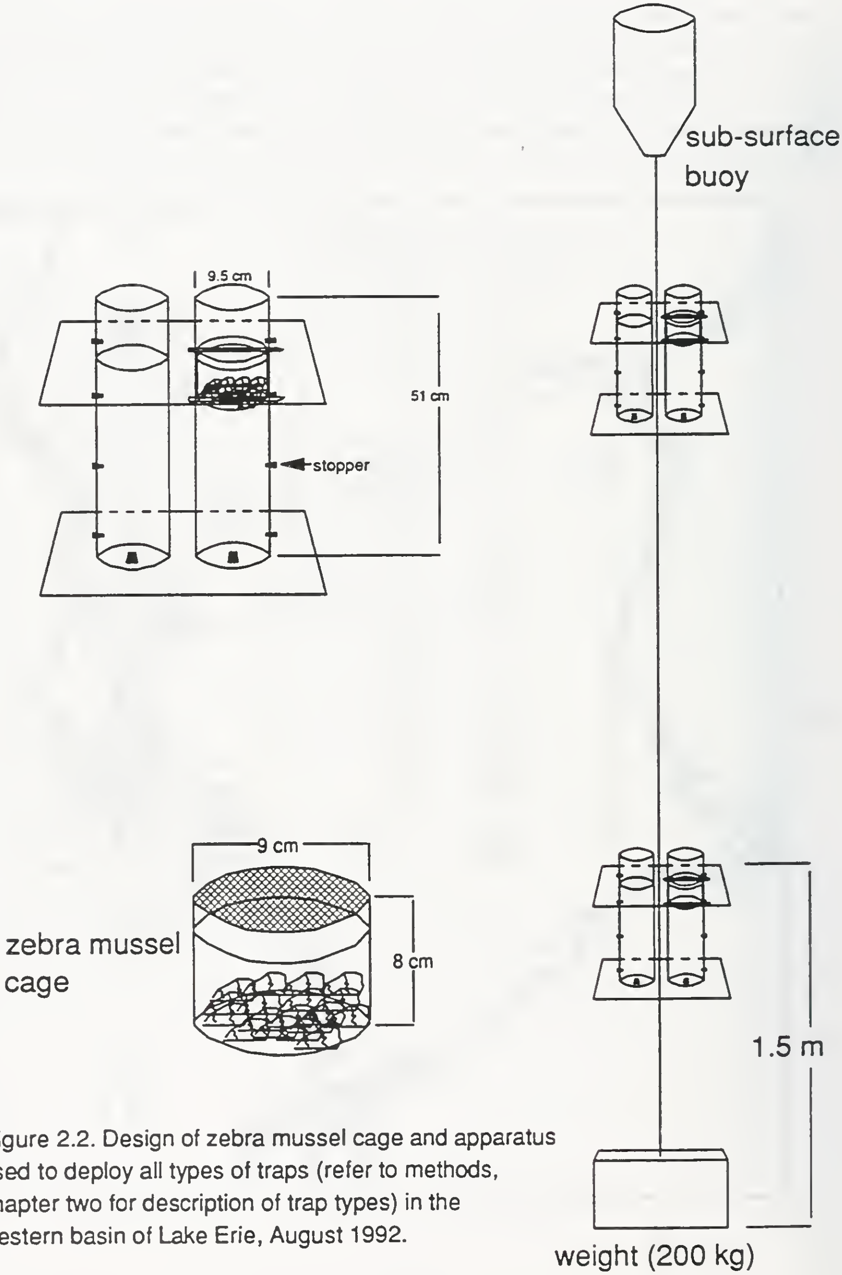


Figure 2.2. Design of zebra mussel cage and apparatus used to deploy all types of traps (refer to methods, chapter two for description of trap types) in the western basin of Lake Erie, August 1992.

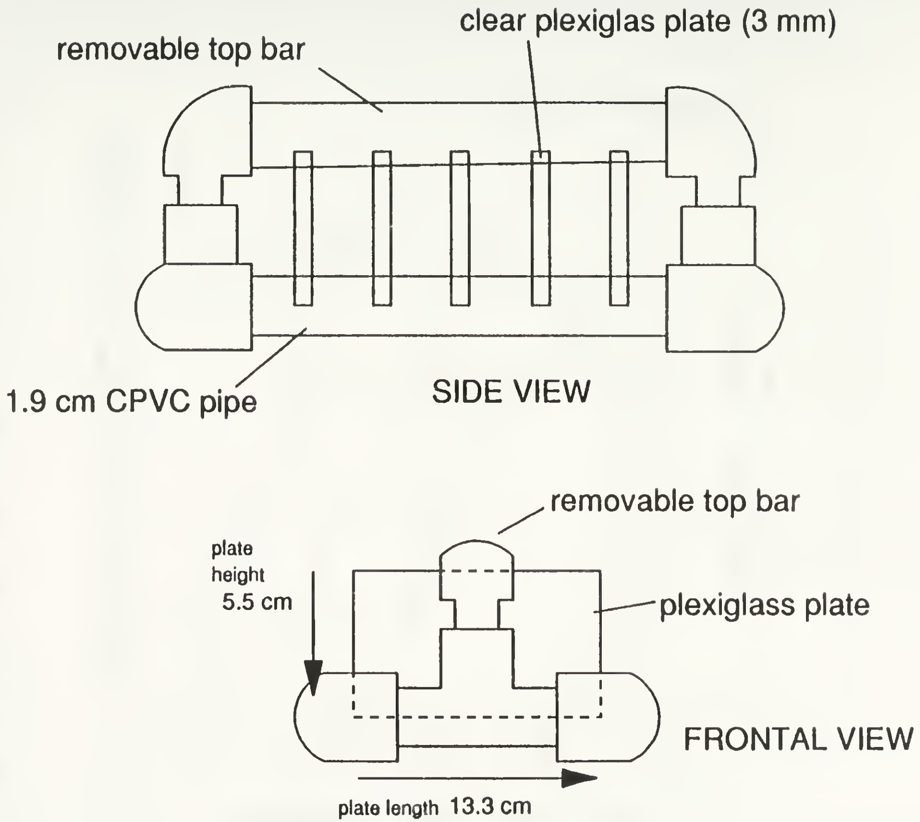


Figure 2.3. Design of settlement racks used in the western basin of Lake Erie for the purpose of determining the recruitment of zebra mussels July to September 1992.

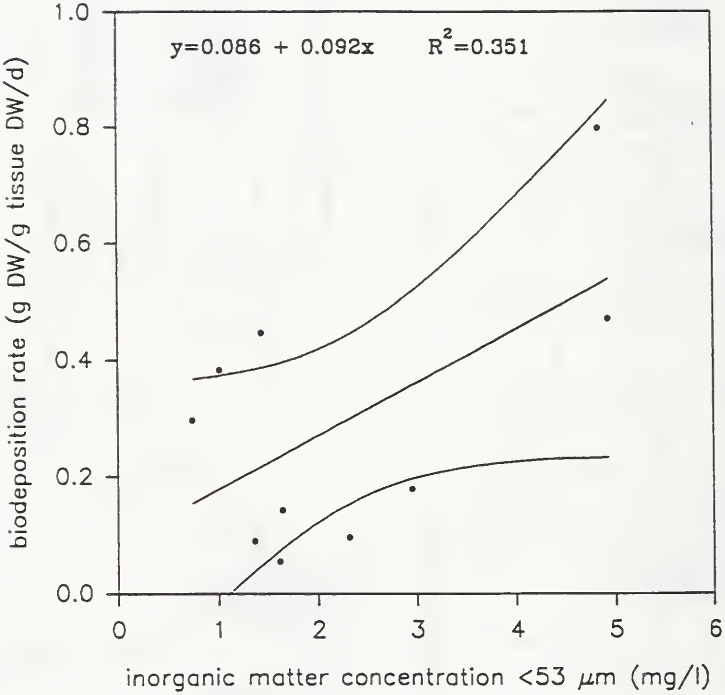
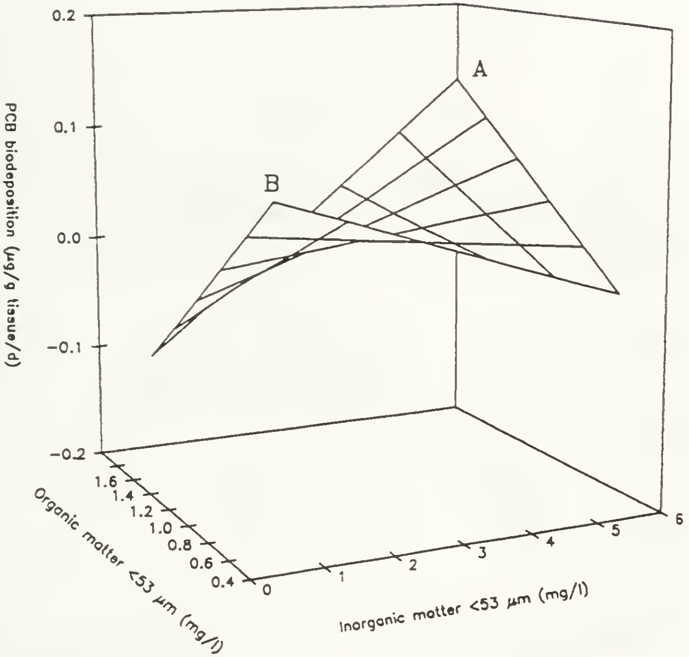
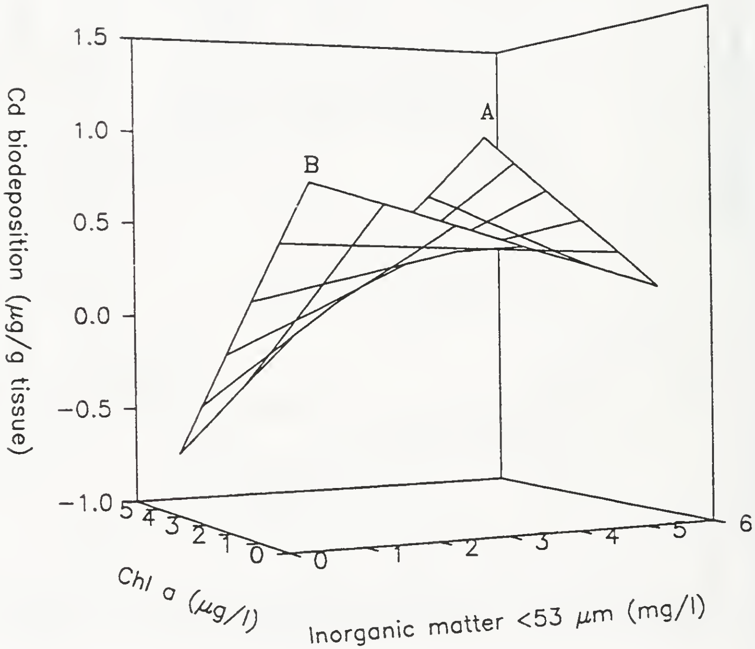


Figure 2.4. Regression of zebra mussel biodeposition rate versus inorganic matter <53 μm (IM) (N=10, dotted lines are 95% confidence bands).



$y=0.176-(0.046)IM-(0.171)OM+(0.53)IM*OM \qquad R^2=0.722$

Figure 2.5. Interaction of organic matter concentration (OM) in the suspended matter and inorganic matter concentration (IM) in the suspended matter on PCB biodeposition rates by zebra mussels in the western basin of Lake Erie, August 1992.



$y=0.844-(0.144)IM-(0.319)chl\ a+(0.089)IM*chl\ a \qquad R^2=0.903$

Figure 2.6. Interaction of inorganic matter concentration (IM) in the suspended matter and chlorophyll a (chl a) concentration in the suspended matter on Cd biodeposition rates by zebra mussels in the western basin of Lake Erie, August 1992.

CHAPTER THREE:

ACCUMULATION OF PCBs AND Cd BY ZEBRA MUSSELS (*Dreissena polymorpha*) IN WESTERN LAKE ERIE AND HAMILTON HARBOUR

3.1 INTRODUCTION

Zebra mussels may potentially be altering contaminant dynamics through accumulation of contaminants in their tissue. The second objective of this study was to assess the ability of zebra mussels to accumulate t-PCBs and Cd in their tissues, and to assess the potential impact of zebra mussels on contaminant dynamics as a consequence of this accumulation. Within this objective are two hypotheses: 1) zebra mussels accumulate significant amounts of t-PCBs and Cd in their tissue. The potential for this accumulation to be high enough to alter contaminant movement will also be investigated; and 2) there is a significant relationship between food quality factors (organic/inorganic matter levels in suspended particulates, chl a) and t-PCBs and Cd uptake.

Zebra mussels can accumulate high levels of PCBs and other organic contaminants (Reeders and Bij de Vaate 1992; Brieger and Hunter 1993), as well as Cd and other metals (Secor et al. 1993; Mersch and Pihan 1994) in their tissues. The prediction of the first hypothesis is that zebra mussels transplanted from an area of low contamination into locations of high contamination will have significantly greater concentrations of t-PCBs and Cd in their tissues after a 3 week exposure period than those initially present.

As filter feeding animals, mussels are particularly susceptible to accumulation of contaminants across the G.I tract after ingestion of contaminated suspended food (Knezovich et al. 1987; Schrap 1991). Specifically, contaminant uptake by zebra mussels via food appears to be an important route of accumulation. Bruner and Fisher (1993) have shown that uptake of contaminated algal particles by mussels is an important exposure route. Brieger and Hunter (1993) observed a lower uptake rate of PCBs from water than from food. For inorganic contaminants, removal of organic matter from sediments was followed by a decrease in the uptake of Cd by Macoma balthica (Harvey and Luoma 1985). Harvey and Luoma (1985) suggest that the decrease in contaminant concentration occurred because the organic source of Cd exposure was removed. Availability of metals was also decreased when iron oxide was the dominant component of the sediment (Harvey and Luoma 1985). Tessier et al. (1984) also saw an inverse relationship between trace metal accumulation in Elliptio complanata and sediment iron oxide concentration. In the present study, since iron oxides are components of inorganic matter, the inorganic matter content of the suspended particulates to which mussels are exposed should decrease the availability of Cd. Given the findings of these studies, the present study will attempt to model the accumulation rate of contaminants by zebra mussels. Therefore, the prediction of the second hypothesis is that accumulation of t-PCBs and Cd by zebra mussels will be positively correlated with the food content of the suspended material, measured as organic matter and chl a, and Cd accumulation will be negatively correlated with inorganic matter concentration in the suspended particulates.

3.2 METHODS AND MATERIAL

3.2.1 Western Basin of Lake Erie Study

In August, 1992, a study was conducted in the western basin of Lake Erie to determine whether zebra mussels accumulate t-PCB and Cd. The potential effects of this on contaminant dynamics is also considered. This study was carried out concurrently with the biodeposition experiment (chapter two), therefore the same sites were used. On August 20, 1992, mussels were collected from the central basin of Lake Erie at Port Stanley, Ontario (Fig. 3.1) and transplanted to all sites in the western basin Lake Erie (Fig. 2.1). The intention here was to remove mussels from a site of low contamination (Port Stanley- PCBs, Simmons 1984; Cd, Lum 1987) into a more contaminated area (western basin of Lake Erie- same references as Port Stanley). Mussels were carefully scraped from rocks approximately 1 m under the surface of the water at Port Stanley and placed in cages constructed of Plexiglas tubing and fibreglass screening (Fig. 3.2). Cages were immediately placed into coolers filled with water from Port Stanley for transportation to the western basin.

All mussels were in place at the western basin sites within 3 h of their removal from Port Stanley. One cage of mussels was placed at each site using SCUBA divers. Cages were wired onto the line at each site at the base of the sediment traps (3.5 m under the surface, 1 m off the bottom). In addition, just prior to placement, three samples of mussels were removed for the purpose of determining the initial tissue concentrations of t-PCBs and Cd. Transplanted mussels were removed from the water on September 12, 1992.

On September 20, 1992, the associated coast guard buoys at each site (except Point Pelee because there was no buoy here) were sampled for existing zebra mussels along the lengths of buoys, chains, and anchors. Length-frequency analysis of the mussels suggested that the mussels were 1.5 to 2 years old. Since most zebra mussels die after 1.5 to 2 years (Mackie 1993), it was assumed that contaminants, if present, would have reached a steady state in these mussels. The theoretical steady state level of a contaminant in an organism is that where the uptake and elimination of the contaminant by the animal are in equilibrium, thus the concentration remains constant (Fletcher and Mackay 1993). Hence, by comparing contaminant levels between transplanted and *in situ* mussels, it could be determined if transplanted mussels had reached a steady state level of the contaminant. Samples of mussels were removed from the buoy and chain at the depth which corresponded with the location of the transplanted mussels. At some sites no mussels were found at these corresponding depths on the buoys and chains. In these cases either no mussel data were collected, or the next closest spot was sampled if mussels were present.

3.2.2 Tissue Removal Procedures

Upon removal of all samples (transplant and *in situ* mussels), mussels were wrapped in hexane-rinsed aluminum foil and placed on ice until return to the laboratory where they were immediately frozen at - 70°C. At a later date, appropriate weights of tissue were separated from the shells of mussels (approximately 5-7 g wet weight for each t-PCB sample, 2 g wet weight for each Cd sample) for analysis. This was done by removing mussels randomly from the sample within an approximate size range of 1-2 cm. All implements used in the tissue removal were made of stainless steel. Prior to the tissue removal for each sample, the implements were thoroughly cleaned with soap and water then hexane-rinsed when used for PCB tissue samples, or acid washed in 10% HCl solution when used for Cd tissue samples. Tissue samples were put into glass vials. Vials used for PCB tissues were cleaned with soap and water, rinsed several times with de-ionized water, oven dried, then hexane-rinsed twice. Caps for vials were lined with hexane-rinsed aluminum foil. Vials used for Cd tissues were cleaned with soap and water, rinsed several times with de-ionized water, and soaked in a 10% HCl acid bath for 24 h, after which they were rinsed thoroughly with de-ionized water and oven dried. Cleaned vials not in use were capped at all times. Once an appropriate tissue weight was collected, the sample was labelled and returned to -70°C until analysis was performed. Four replicates of mussel samples were used for each site for both t-PCB and Cd transplants, and three replicates of mussels were used from *in situ* samples. Prior to any tissue removal period, the immediate work area was washed down with soap and water. With each sample, tissue was removed from shells on a new surface of aluminum foil to minimize any potential contamination from the table top surface.

All shells were kept after the tissues were removal. Each sample of shells was digitized using a Hipad™ digitizer pad and a computer program developed by Mr. Russel Hopcroft, University of Guelph. The data were used to determine the length-frequency distribution and average length of each sample so that size comparisons could be made among all samples.

3.2.3 Contaminant Analysis of the Tissues

All contaminant tissue analyses were carried out by Ortech Corporation, Mississauga, Ontario. The samples were analyzed by the methodology used by the Ontario Ministry of the Environment and Energy for fish tissue analyses. For t-PCB analysis, approximately 5 g of wet tissue for each sample was homogenized and acid digested with concentrated HCl. The digested material was then double extracted with 25% methylene chloride in hexane and the solution was neutralized with $NaHCO_3$. The sample was brought to a 100 ml volume using the solvent mixture and aliquots were taken for t-PCB and lipid content analyses. Cleanup on PCB extracts was carried out by 2% water-deactivated Florisil, and were then analyzed by dual capillary GC/ECD. The PCB reference used in the analysis was Aroclor 1254/1260 (4/1). The solvent in lipid aliquots was allowed to evaporate by air-drying and lipid content was then determined gravimetrically. Moisture content of each sample was also determined gravimetrically.

For cadmium analysis, approximately 2 g of wet tissue per sample was used. Moisture content of each sample was determined by drying at 100°C in a pre-weighed flask. The sample was then diluted twice with 50% nitric acid, and then again with concentrated perchloric acid. The digested solution was diluted to 4% solution of HCl acid. Analysis for Cd was done using flame atomic absorption spectrophotometry.

Several procedures were carried out during all analyses to ensure proper QA/QC. For PCB determinations, fish samples spiked with known amounts of PCBs were included in the analysis which gave 95% recovery. As well, several samples were measured in duplicate and the values of duplicates were similar to each other (Table 3.1). For Cd, spiked samples were also included and gave a range of recoveries from 76 to 98.6 percent (Table 3.2). Also for Cd, an oyster tissue standard (SRM 1566) was analyzed which gave results well within the certified value (Table 3.3).

Environmental factor data collected during the biodeposition study (chapter 2) were also used in the analysis for the accumulation experiment (this chapter). The use of these data in relation to rates of accumulation of PCBs and Cd by mussels are described later.

3.2.4 Hamilton Harbour

Coinciding with the experiment of objective one, a contaminant accumulation experiment was also carried out in Hamilton Harbour. Mussels were transplanted from the central basin of Lake Erie at Port Stanley, Ontario to Hamilton Harbour on September 29, 1992. The same procedures as described above were used. Mussels were placed off the pier at C.C.I.W. at the two depths within 3 h of collection from Port Stanley. These depths were 1.5 m and 9 m, in 9.5 m of water. These were left in place until November 11, 1992. Upon collection mussels were placed on ice until being brought back to the laboratory where they were immediately frozen at -70°C. Tissue samples for both the western basin and Hamilton Harbour were submitted for t-PCB and Cd analyses to Ortech Corporation at the same time. The procedures were the same as previously described.

3.2.5 Gut Evacuation Experiment

Throughout the study mussels were not allowed to evacuate gut contents upon collection due to logistical constraints. It has been suggested that the apparent contaminant concentration in tissue of organisms can be elevated because of the possible contamination of gut material (Luoma 1983). Gut contents have been accounted for in some studies (Streit and Winter 1993, Tessier et al. 1993). Other studies did not allow for gut evacuation in test organisms (Secor et al. 1993, Reeders and Bij de Vaate 1992). To determine the importance of gut evacuation for t-PCB and Cd body burdens in zebra mussels, a laboratory study was conducted.

On November 22, 1993 mussels were collected from the field at Colchester, Ontario, on the western basin of Lake Erie. Water temperature was 8°C at the lake. Therefore mussels were placed in lake water in the laboratory for a 24 h period, with the intention of acclimating them to room temperature (20°C). All mussels were placed in 1-L glass flasks which were previously cleaned with soap and water, and hexane-rinsed or acid washed (with 10% HCl solution), depending on the contaminant that the sample was used for. Three initial mussel samples for both t-PCB and Cd analysis were taken at the beginning of the experiment. After 24 h, mussels were placed in 1:1 well water/de-ionized water for another 24 h. By mixing well water and de-ionized water in this ratio mussels are then exposed to water hardness levels similar to those of Lake Erie. Three mussel samples for both t-PCB and Cd analyses were removed at the end of this time in order to observe any difference in contaminant concentrations from that of the starting samples. The content of the guts were not checked to see if mussels had completely evacuated. All samples, both whole mussel and tissue, were frozen at -70°C prior to analysis. Samples were analyzed by Ortech Corporation for t-PCBs and Cd using the methods previously described.

3.2.6 Calculations and Statistical Analysis

Where appropriate (see results, this chapter) PCB concentrations in mussel tissue were corrected for lipid content. This was done by dividing the whole tissue concentration of PCBs by the lipid content for each sample. Otherwise, PCB concentrations are reported on a dry weight basis.

To determine if mussels transplanted into the western basin of Lake Erie accumulated significant amounts of PCBs and Cd, all samples were compared to the initial Port Stanley level using an ANOVA ($p \leq 0.05$) (Snedecor and Cochran 1989). Dunnett's test ($p \leq 0.05$) was then used to determine any statistical differences between sites and Port Stanley to determine if transplanted mussels reached a tissue level similar to the *in situ* mussels, a standard two-sided t-test was performed ($p \leq 0.05$).

As an estimate of the potential impact of mussels on contaminant dynamics through tissue uptake, contaminant tissue levels were calculated on a per unit area basis. This was done by calculating the weight distribution of the mussels in the transplant tissue samples using the equation $y = 0.007x^{2.982}$ (Mackie 1993) where 'x' is the shell length and 'y' is the dry tissue weight. This weight was then multiplied by the number of mussels per m^2 determined from the mussels found on the settlement racks on the last day of the sampling period (September 12, 1992), and then divided by the number of mussels digitized in the transplant sample. This calculated weight per m^2 was then multiplied by the mean tissue concentration of the contaminant in the transplanted mussels at the given site. The following assumptions were made in order to do this calculation:

1) The weight distribution of mussels in a transplant sample was representative of those in a typical weight-frequency distribution curve despite this not including small mussels (< 1 cm) and large mussels (> 2 cm), since these sizes were deliberately not included in the tissue sample.

2) The number of mussels per m^2 calculated from rack data on September 12, 1992, was representative of the natural density of mussels found at that site. As previously mentioned, this may be an over estimate of natural densities at sites, however it is representative of mussel densities found on solid substrates in the western basin of Lake Erie.

Accumulation rates of PCBs and Cd by the transplanted mussels were determined by subtracting the average initial contaminant concentration (Port Stanley mussels) from each of the tissue levels of the contaminant found in the transplanted mussel samples after deployment, then dividing this number by the number of deployment days. This is in fact not a true rate since uptake is generally not linear over the accumulation phase in an animal. However, this calculation is used here since the 'rate' will be relative among sites, and serves as a useful means of comparison. From herein 'rate(s)' refers to the above method of calculation, with the above limitations of this measurement. Accumulation rates were correlated with environmental variable data using multiple regressions (SAS 1987). The level of acceptance was set at $p \leq 0.05$.

3.3 RESULTS

3.3.1 Gut Evacuation Experiment

The results from the gut evacuation experiment are presented in Table 3.4. There was no significant difference (student t-test, $p>0.05$) between the initial tissue concentration and the 48 h tissue concentration for both the PCBs and Cd.

3.3.2 PCB Accumulation

The average length of shells of zebra mussels in each transplant sample submitted for PCB analysis is presented in the appendix IX. Values of shell length all fell within the chosen size range of 1 to 2 cm.

Some significant differences (ANOVA, $p<0.05$) existed between lipid levels of the initial Port Stanley mussels and mussels collected after transplanting in the western basin and Hamilton Harbour. In addition, significant differences ($p<0.05$) existed in lipid content of transplanted mussels and *in situ* mussels in the western basin. Therefore, all PCB tissue levels for these comparisons are presented on a lipid corrected basis. Average lipid values and indications of variability for experimental mussels are presented in Appendix X.

Concentrations of PCBs in mussels transplanted to Hamilton Harbour at both top and bottom sites were significantly greater than those in the initial tissue levels of Port

Stanley mussels (student t-test, $p < 0.05$). The levels (lipid corrected) were as follows: top site, 170.9 ± 17.0 ng PCBs/g lipid (x 100): bottom site, 134.3 ± 11.9 ng PCBs/g lipid (x 100); initial site, 13.3 ± 3.5 ng PCBs/g lipid (x 100).

Concentrations of PCBs in transplanted mussels in the western basin of Lake Erie were all significantly greater than the Port Stanley initial sample (Dunnett test, $p < 0.05$) (Fig. 3.3).

In order to determine if the transplanted zebra mussels had reached the same concentration of the contaminant as the *in situ* mussels, transplanted samples were compared with *in situ* samples which were assumed to have reached steady state at a given site. When comparing these samples, there was no significant difference at sites E10 bottom, and H top and bottom. At sites E13 top and bottom, *in situ* samples were significantly lower than the transplant samples (student t-test, $p < 0.05$) (Fig. 3.4). At Pelee Island bottom site there was a weakly significantly lower level in the *in situ* sample than the transplant ($p = 0.059$) (Fig. 3.4). Data were not normal and different transformations were performed (log, natural log, log ± x). Despite this, normalization of the data was not possible, although log transformations gave the best results.

Tissue PCB accumulation rates on a per m^2 basis were estimated for each site in the western basin, except at Point Pelee (no growth racks were used here) by using methods described previously (see methods and materials). Results for this are presented in Table 3.5. Values varied considerably from site to site. The average value for all sites in the western basin was 187923 ± 60850 $ng/m^2/d$. No calculations of this nature were possible for Hamilton Harbour since settlement data were not available.

3.3.3 Cadmium Accumulation

The average length of shells of zebra mussels in each transplant sample submitted for contaminant analysis is presented in the appendix XI. All values fell within the desired size range of 1 to 2 cm.

Cadmium tissue levels in mussels transplanted into Hamilton Harbour were significantly lower than the initial tissue levels for Port Stanley (student t-test, $p<0.05$). The levels are as follows: top site, 1.81 ± 0.23 μg Cd/g DW tissue, bottom site, 2.00 ± 0.43 μg Cd/g DW tissue, initial site, 3.25 ± 0.27 μg Cd/g DW tissue.

Concentration of Cd in transplanted mussels in the western basin sites were all significantly greater than those in the initial samples from Port Stanley, with the exception of Pelee Island bottom and Point Pelee bottom sites, which were not statistically different from each other (Dunnett Test, $p>0.05$) (Fig. 3.5).

When comparing transplants with *in situ* mussels for Cd tissue levels, all *in situ* samples were significantly greater ($p<0.05$) than the transplant samples, with the exception of site H bottom, where there was no difference in Cd tissue levels (student t-test, $p>0.05$) (Fig. 3.6). All data met the tests for normality (using stem-leaf plots, box plots, skewness and kurtosis values).

Table 3.6 presents the estimated tissue accumulation rates per m^2 for Cd for all locations in the western basin except Point Pelee. Values varied considerably from site to site. The average value for all sites in the western basin was 1209 ± 274 ug/m^2/d. No estimate of this for Hamilton Harbour was possible since settlement data was not available.

3.3.4 PCB Accumulation Model

Accumulation rates of PCBs per unit dry weight mussel tissue for sites in the western basin and Hamilton Harbour are presented in Table 3.7. PCB accumulation rates for the western basin were regressed against environmental factors (Appendix VI) through multiple linear regression analysis (SAS 1987). The best fitting model incorporated the factor inorganic matter $<53\ \mu m$ (IM) ($p=0.021$) (Appendix XII). The equation is:

$$y = 5.924 - (0.798)IM \qquad R^2=0.507$$

Normality tests (stem-leaf plots, box plots, skewness and kurtosis values) showed data to be normally distributed.

Graphical representation of this model is presented in Figure 3.7. PCB accumulation showed a negative relationship to inorganic matter concentration $<53\ \mu m$.

3.3.5 Cadmium Accumulation Model

Estimates of accumulation rates of Cd per unit dry weight of mussel tissue in the western basin of Lake Erie and in Hamilton Harbour sites are presented in Table 3.8. Cadmium accumulation rates for the western basin were regressed against environmental factors (Appendix VI) through multiple regression analysis (SAS 1987). The best fitting model incorporated the factors inorganic matter $<53\ \mu m$ (IM), organic matter $<53\ \mu m$ (OM), and chl $\underline{a}$ ($p=0.017$) (Appendix XIII). The equation is:

$$y = 0.15 - (0.08)IM + (0.51)\,OM - (0.05)Chl\ \underline{a} \qquad R^2=0.796$$

Normality tests (stem-leaf plots, box plots, skewness and kurtosis values) showed data to be normally distributed.

Graphical representation of this model is presented in Figure 3.8. When keeping organic matter constant in the regression model at either a low, medium, or high level an overall trend of increasing accumulation rates of Cd occurs as the level of organic matter increases from the low to high levels. The lowest accumulation rate Cd is seen with high levels chl a and inorganic matter, but low levels of organic matter. The highest accumulation rate of Cd is seen with low levels of chl a and inorganic matter, but high levels of organic matter.

3.4 DISCUSSION

3.4.1 Gut Evacuation Study

The results from the gut evacuation study appear to show that there was no effect of gut content on contaminant tissue burdens over a 48 h period. This is in contrast to other studies (Flegal and Martin 1977, Elwood et al. 1976) who found that gut contents can add to the apparent tissue level of the contaminant in small invertebrates. In the present study it was not practical to allow for depuration of animals after collection because of the number of samples and each sample consisted of up to several hundred individuals. Results here may have differed from other studies because mussels may not have fully evacuated contaminated gut material after removal (48 h), perhaps due to acclimation conditions. The acclimation period (i.e. the initial lake water exposure in the first 24 h) to room temperature (20°C) may not have been a long enough adjustment time since the water temperature that the mussels were removed from was quite low (8°C). As well, checking of gut contents upon completion of the experiment was not done. This would have been appropriate in order to verify evacuation. However, mussels did appear to be siphoning during the entire experiment, which would imply that they were feeding/evacuating during this period.

3.4.2 PCB Accumulation

The results from accumulation studies are in agreement with the first prediction of objective two that zebra mussels transplanted into the western basin of Lake Erie will accumulate significantly greater amounts of contaminants compared to the initial level.

In the case of PCBs, concentrations at all sites in the western basin and Hamilton Harbour were significantly greater than the initial concentration of PCBs in mussels from Port Stanley, Ontario. PCBs, being lipophilic, accumulate relatively quickly in tissues. Kauss et al. (1981) observed a rapid increase of PCB levels in tissues of Elliptio complanata after only 2 days in the Niagara River. Other studies observed significantly elevated concentrations of organic contaminants in the tissues of E. complanata after 3 weeks of exposure in the Detroit River (Kauss and Hamdy 1985) and the St. Clair River (Muncaster et al. 1989). Zebra mussels also accumulate organic contaminants quickly in the laboratory (Brieger and Hunter 1993). The results of the present study verify that zebra mussels will accumulate PCBs rapidly under field conditions as well.

Given that zebra mussels can accumulate significant levels of PCBs, and that they are found at very high densities in the Great Lakes, the importance of tissue accumulation of contaminants on contaminant dynamics should be considered. The amount and rate of contaminant accumulation are important points to consider. The theoretical steady state level of a contaminant in an organism is that where the uptake and elimination of the contaminant by the animal are in equilibrium (Fletcher and McKay 1993), thus tissue concentrations remain constant. In the present study, contaminant levels in tissues of transplant mussels were compared with those of *in situ* mussels, which were assumed to be at a steady state for the contaminant. The results show that after the 21 day exposure period transplant mussels had accumulated either more or equal amounts of PCBs compared to the *in situ* mussels. Several studies with other molluscs have shown varying results with respect to the amount of time required to reach a steady state in organic contaminant tissue concentration. Muncaster et al. (1989) observed an equilibrium between body burdens of hexachlorobenzene and ambient concentrations in 21 days. Kauss et al. (1981) observed equilibrium levels of PCBs in clams and water after 16 days in the Niagara River. Under laboratory conditions Brieger and Hunter (1993) saw maximum levels of various PCB congeners introduced to zebra mussels through food after 10-21 days. In the present study it appears that 21 days exposure allows for accumulation of PCBs to levels as high as or greater than *in situ* mussels.

Higher levels of PCBs found in the transplanted mussels compared to the *in situ* mussels at some sites could be due to the lipophilic nature of PCBs. The steady states of many organic contaminants in tissues may require an extended period of time to be reached because an animal's elimination processes 'lag' behind uptake processes (Oliver 1984, Oliver and Niimi 1983). Perhaps the levels observed in the *in situ* mussels are only reached after a period longer than 21 days at these sites. Prior to 21 days there is a greater accumulation of PCBs than the actual steady state level since the elimination processes may not yet have equilibrated with the uptake processes for the contaminant. As a result, uptake of the contaminant would continue to increase above the steady state level, then decrease as equilibrium is attained.

The above results establish that zebra mussels can contain significant levels of PCBs. Tissue levels of PCBs in the present study were often lower than those found in other studies. Brieger and Hunter (1993) saw a range of PCB concentrations (measured as Aroclor 1242 and 1254) from various western basin sites of 154 ng/g wet weight to 341 ng/g wet weight in mussels, with an average of 228.3 ng/g. Kreis et al. (1991) reported t-PCB concentrations of 520 ng/g wet weight in western basin zebra mussels. In the present study *in situ* samples ranged from 25.6 to 209.5 ng/g wet weight, with an average of 104.9 ng/g. These slightly lower values could be due to the location of collection which was not the same as these other studies. None-the-less, these values are of the same order of magnitude as those in other studies.

Because of the high natural densities of mussels which exist in the Great Lakes, and the high tissue burdens of contaminants in these mussels, contaminant movement through the water column could be affected by the tissue uptake of these contaminants. Calculated rates of uptake on a per unit area basis are extremely high. The potential importance of contaminant accumulation by zebra mussels on contaminant dynamics has been suggested by other researchers (Bruner et al. 1991). Estimates of PCB tissue burdens of zebra mussels on a lake wide basis were calculated by Brieger and Hunter (1993) for Lake St. Clair; their value was estimated to be 280 kg of PCBs held in the tissues of mussels in this lake. A similar value can be calculated for this study. Assuming the bottom of the western basin of Lake Erie is at least equal to the surface area (3080 km^2, Mortimer 1987) and that the average tissue concentration (based on data from the present study, mean tissue accumulation of the transplanted mussels) of 615.9 ng/g dry weight, with a mussel density of approximately 6.8 kg/m^2 tissue dry weight (also from this study), then mussel tissues are holding 12880 kg of PCBs. The daily input of t-PCBs into western Lake Erie from the Detroit River is estimated at 0.5 kg/d (MDNR and OMOE 1991). My estimate of the basin wide PCB tissue burden is very high in relation to this input value. It should be noted that my estimate of the PCB tissue burden is very rough, limited in extrapolation by, among other things, the time of year and the accuracy of the parameter values (i.e. lake wide density of mussels and PCB tissue concentration). My estimates of mussel densities may infact be too high to be considered representative of the average basin value. The contaminant tissue burden estimate is based on the sites used in the present study only and they probably do not represent all locations. As discussed in chapter two, the density estimates calculated from the settlement rack data are probably only representative of areas of the western basin where there are solid substrates for mussels to settle on, such as shoals. Extrapolating the densities calculated in the present study to the entire basin may result in an elevated number for the amount of contaminant held in the tissues. None-the-less, the estimate of PCB tissue burden for the basin shows that zebra mussels have the potential to alter the movement of PCBs during lake cycling processes by storing these contaminants in tissues.

If mussels are altering contaminant movement by uptake, the tissue release of these contaminants by dead and decaying animals should be considered. The present study did not consider this aspect, however, other studies have assessed the leaching of contaminants from zebra mussel tissue. Doherty et al. (1993) did not show any potential leaching risk of PCBs by mussels (based on E.P.A. leaching assessment tests), although tissue levels in collected mussels were quite low compared to the present study. Contaminant release upon death through decay for mussels in areas of high contamination should be an important consideration in future studies dealing with the effects of accumulation on contaminant movements.

3.4.3 Cadmium Accumulation

Cadmium concentrations in zebra mussels transplanted into the western basin of Lake Erie were significantly greater than the initial concentration in mussels from Port Stanley, Ontario, with the exception of Pelee Island bottom and Point Pelee bottom where there were no significant differences found. Cd, like most metals, does not accumulate in the tissues of biota as rapidly as organic chemicals (for example see Kauss et al. 1981). In the present study accumulation of Cd appears to be significant, as shown by differences in concentrations between transplanted mussels and the initial sample for many of the sites, but does not accumulate as readily as PCBs in the tissues. This is shown by the Cd tissue comparisons between the transplanted mussels and the *in situ* samples. Here, most locations had higher *in situ* levels of Cd than those for the transplanted mussels. Van Hattum et al. (1989) did not see Asellus aquaticus reach a levelling off point in the tissues after 35 days of exposure to Cd in food. Chr de Kock and Bowmer (1993) did not observe a steady state of Cd in zebra mussels in field experiments until 40-60 days. Therefore, the exposure time of transplants in the present study was probably not long enough for the mussels to reach a steady state of the chemical, although significant levels of Cd can still be accumulated.

Despite the lower Cd tissue levels in transplanted mussels compared to *in situ* mussels, tissue levels of these contaminant in the transplanted mussels may be high enough to alter Cd movement in the western basin of Lake Erie. Using the same parameter estimates used with PCBs and an average tissue Cd concentration of 3.6 μg/g dry weight, the amount of Cd accumulated in the tissues of mussels in the western basin is estimated at 75398 kg. This is a high value considering that the estimated inputs from the Detroit River into the western basin is 18 kg/d (MDNR and OMOE 1991). As with the PCB calculation, this tissue burden estimate for Cd is only approximate, but can be useful in showing that mussels may have an important role in the movement Cd during lake cycling processes.

The results for Hamilton Harbour for Cd are interesting in that transplanted mussels actually decreased their contaminant tissue levels compared to the initial levels. This is not easily explained since metal levels in Hamilton Harbour were quite high in the sedimented material (maximum of 3.00 ± 0.19 μg/g DW sediment) compared to western basin sites (maximum of 0.96 ± μg/g DW sediment). In addition, at Hamilton Harbour sites high levels of PCBs were accumulated in zebra mussel transplants relative to the initial levels (eg. 170.0 ± 17.0 ng/g for the top site versus 13.3 ± 3.5 ng/g for the Port Stanley initial sample), implying that mussels were filtering and feeding on contaminated material. One possible explanation for the decrease in Cd tissue levels is that mussels may be biodepositing all Cd which passes through their bodies. If at this time of year (fall) there were relatively high levels of suspended inorganic matter (Jaramillo et al. 1992), then perhaps mussels deposited this as pseudofeces. The biodeposit concentration of Cd was quite high relative to the sediment values (eg. 8.92 ± 3.38 μg/g DW biodeposits versus 2.50 ± 0.21 μg/g DW sediments), suggesting that Cd-laden inorganic particulates (e.g. iron oxides) in pseudofeces may be the reason for the high levels here. In conjunction with mussels biodepositing the Cd, mussels may have eliminated some of the residue tissue Cd, thus resulting in negative accumulation rates.

3.4.4 PCB Accumulation Model

PCB accumulation by zebra mussels was negatively correlated with inorganic matter, therefore results from the PCB accumulation model did not agree with the prediction that organic matter and chl a would be positively correlated with uptake of PCBs. However, results can be explained from the literature.

The negative relationship between PCB accumulation and suspended inorganic matter content may be due to a decrease in the ingestion rate of food. This is supported by the positive relationship between biodeposition and inorganic matter (chapter two), since increasing suspended inorganic matter content may be increasing pseudofecal production and decreasing ingestion rates of particulates (Foster-Smith 1975). This would then lead to less food intake and less contaminant accumulation.

3.4.5 Cadmium Accumulation Model

The results from the Cd accumulation model are in partial agreement with the prediction that Cd accumulation in zebra mussels will be positively correlated with organic matter and chl a levels, and negatively correlated with inorganic matter levels.

The lowest accumulation was observed in the model when chl a and inorganic levels were high, and organic matter levels were low. Cd sorbs, to a large extent, onto inorganic particulates such as iron and manganese oxides (Tessier et al. 1993). Past studies have shown that high iron oxide content in sediments will decrease metal accumulation in molluscs (Tessier et al. 1984, Harvey and Luoma 1985). The low tissue accumulation rate of Cd observed in the present study may be due to the high inorganic matter content of suspended particulates (i.e. high iron oxide content) which would decrease the availability of Cd to the mussel.

The high chl a and low organic matter levels associated with the low accumulation rate is interesting. Harvey and Luoma (1985) showed that Macoma balthica in sediments with low organic matter content had low levels of Cd. They suggest that the food route of exposure was removed under these circumstances. The present study supports this since at low organic matter levels the accumulation rate of Cd was also low. However one would expect the relationship between chl a content and Cd accumulation to be similar to the relationship between organic matter and Cd accumulation because algae is a large component of organic matter. It may be that mussels reduce ingestion rates since food availability would be high and plentiful during conditions of high chl a. Sprung and Rose (1988) also observed a decrease in ingestion rate at high algal concentrations in the laboratory. Reduced ingestion rates, in turn, may reduce accumulation of Cd.

The highest accumulation rates of Cd by mussels was seen when chl a and inorganic matter levels were low, while organic matter levels were high. This is in agreement with the above discussion. Mussels may increase their ingestion rate when chl a levels are low. Given that organic matter levels are high at this same time, thus allowing for potentially more available Cd (Harvey and Luoma 1985), then the trend of higher Cd accumulation is expected.

3.5 CONCLUSIONS AND POTENTIAL ECOLOGICAL IMPLICATIONS

This study suggests that zebra mussels may have the potential to alter t-PCB and Cd dynamics through biodeposition and tissue accumulation of these contaminants. This may impact the lake ecosystem as a whole by changing the food chain transfer patterns of these contaminants, as well as their physical fate during cycling.

Natural sedimentation from the water column will be increased by many times in the presence of zebra mussels. This is particularly true in locations where mussel density is high (for example the western basin of Lake Erie). Data presented in chapter two show that biodeposition by mussels is most influenced by the presence of suspended inorganic matter content, where there is a weakly significant ($p < 0.10$) positive relationship between the two.

Biodeposits and sedimented material have similar contaminant concentrations, therefore mussels do not increase or decrease the contaminant concentration of the biodeposits (i.e. through physiological or mechanical processes). Zebra mussels biodeposit PCBs and Cd at a much faster rate than sedimentation processes alone, particularly when compared on a per unit area basis. In the case of PCB biodeposition rates, organic and inorganic matter levels in suspension have the greatest influence. The rates of Cd biodeposition is significantly ($p < 0.05$) related to levels of chlorophyll a and suspended inorganic matter.

Zebra mussels can accumulate significant levels (approaching or exceeding steady state) of PCBs and Cd in their tissues in a relatively short period of time (21 d). This may be particularly important with respect to the amount of contaminant accumulation in tissues on a lake-wide basis. Because of the high densities of mussels found in the Great Lakes, and that mussel tissue can accumulate high levels of contaminants, the total biomass of mussel tissue may be a major storage point during the cycling of these contaminants. In the case of PCB accumulation, suspended inorganic matter level has a significant ($p < 0.05$) influence on the extent of accumulation. Cd accumulation is significantly related ($p < 0.05$) to inorganic and organic matter levels, and chlorophyll a levels.

The combined effects of biodeposition and accumulation of contaminants by zebra mussels could alter contaminant movements by increasing the removal rate of contaminants from the water column and thus increasing the loads of these to benthic areas. This in turn could have important ecological consequences in that benthic food webs are potentially exposed to more contaminant per unit area (as biodeposition and tissue sources) and/or to a more available form of it. Transfer of these contaminants to pelagic food webs may also be increased through traditional coupling vectors, such as predators of benthic fauna. Consumption of zebra mussels themselves may also be an important transfer mechanism in this respect.

In order to properly assess the importance of zebra mussels in contaminant cycling, future studies must incorporate resuspension factors, as well as tissue decomposition. These are two important means by which contaminants can be reintroduced back into the cycling process. The present study only examined the effects of zebra mussels on the flux of these contaminants through biodeposition, as well as through tissue accumulation of these contaminants. Knowing the extent of resuspension and decomposition processes is critically important in order to better understand the overall effects of zebra mussels on contaminant dynamics in aquatic environments.

Table 3.1. Duplicate PCB concentrations of zebra mussel tissue samples transplanted from Port Stanley, Ontario, to various sites in the western basin of Lake Erie, August 1992. Duplicate measurements were performed as part of quality assurance and control procedures. 'Top' refers to sites where transplanted mussels were placed 3.5 m under the surface of the water, and 'bottom' refers to sites where transplanted mussels were placed 1.0 m off the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	PCB concentration (ng PCBs/g wet tissue weight)
Pelee Island Top replicate #2	123.4 118.4
E13 Top replicate #3	106.9 110.0
H Bottom replicate #3	167.2 127.7

Table 3.2. The percent recovery values of cadmium introduced into various zebra mussel tissue samples from the western basin of Lake Erie. 'Top' refers to sites where zebra mussels samples were placed 3.5 m under the surface of the water, and 'bottom' refers to sites where zebra mussel samples were placed 1.0 m off the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

sample where cadmium was added	**Cd % recovery**
E10 top replicate #1	94.8
E13 bottom replicate #2	93.6
Pelee Is. bottom replicate #2	94.4
E bottom replicate #2	93.5
H top replicate # 2	76.0
Hamilton Harbour initial sample replicate #1	98.6

Table 3.3. Cadmium oyster tissue standard (Standard Reference Material (SRM) lot # 1566, National Institute of Standards and Technology) reporting the certification value and actual values found by Ortech Corporation, Mississauga, Ontario, as part of quality assurance and control procedures.

Oyster Tissue standard (SRM 1566) certification value (μg Cd/g DW tissue)	Value found (μg Cd/g DW tissue)
4.15 ± 0.38	4.25
4.15 ± 0.38	4.08
4.15 ± 0.38	4.25

Table 3.4. PCB and Cd concentrations on the tissues of zebra mussels collected from the western basin of Lake Erie (Colchester, Ontario), November 1993. Mussels were used in a 48 hr gut evacuation experiment. Initial concentrations were at time zero.

sample	**initial concentration (mean ± 1 S.D.)**	**concentration at 48h (mean ± 1 S.D.)**
PCB (ng PCBs/g DW tissue)	166.2 ± 19.3	162.9 ± 43.5
Cadmium (ug Cd/g DW tissue)	3.91 ± 0.90	5.02 ± 0.16

Table 3.5. Calculated accumulation rates of PCBs in zebra mussel tissue per unit area (/m^2) in the western basin of Lake Erie. Zebra mussels were transplanted from Port Stanley, Ontario, into the western basin during August 1992. 'Top' refers to sites where transplanted mussels were placed 3.5 m under the surface of the water, and 'bottom' refers to sites where transplanted mussels were placed 1.0 m off the bottom of the lake. refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	**PCB accumulation rate (ng PCBs/m^2/d) (mean ± 1 S.D)**
E13 top	515729 ± 141263
E13 bottom	304601 ± 82768
Pelee Island top	161837 ± 40247
Pelee Island bottom	75933 ± 25308
E10 top	125547 ± 32351
E10 bottom	125778 ± 114324
H top	98635 ± 30660
H bottom	95325 ± 19878

Table 3.6. Accumulation rates of Cd in zebra mussel tissue per unit area ($/m^2$) in the western basin of Lake Erie. Zebra mussels were transplanted from Port Stanley, Ontario, into the western basin during August 1992. 'Top' refers to sites where transplanted mussels were placed 3.5 m under the surface of the water, and 'bottom' refers to sites where transplanted mussels were placed 1.0 m off the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	**Cd accumulation (μg Cd/m^2/d) (mean ± 1 S.D.)**
E13 top	3759 ± 563
E13 bottom	2010 ± 556
Pelee Island top	366 ± 140
Pelee Island bottom	266 ± 154
E10 top	377 ± 86
E10 bottom	641 ± 160
H top	1040 ± 175
H bottom	1210 ± 355

Table 3.7. Accumulation rates of PCB (lipid corrected) per unit weight of dry zebra mussel tissue in the western basin of Lake Erie, and in Hamilton Harbour. Zebra mussels were transplanted from Port Stanley, Ontario, into the western basin during August 1992, and into Hamilton Harbour in September 1992. Unless otherwise stated, 'top' refers to sites where transplanted mussels were placed 3.5 m under the surface of the water, and 'bottom' refers to transplanted mussels placed 1.5 m off the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	**PCB accumulation rate (ng PCBs/g lipid/d) x 100 (mean ± 1 S.D.)**
E13 top	4.03 ± 0.18
E13 bottom	5.70 ± 0.25
Pelee Island top	5.07 ± 0.22
Pelee Island bottom	3.57 ± 0.16
Point Pelee top	0.64 ± 0.15
Point Pelee bottom	0.57 ± 0.08
E10 top	3.97 ± 0.17
E10 bottom	2.28 ± 0.10
H top	2.76 ± 0.12
H bottom	3.14 ± 0.14
Hamilton Harbour top (1 m below the surface)	6.85 ± 0.74
Hamilton Harbour bottom (8.5 m below the surface)	5.26 ± 0.52

Table 3.8. Accumulation rates of Cd per unit weight of dry zebra mussel tissue in the western basin of Lake Erie, and in Hamilton Harbour. Zebra mussels were transplanted from Port Stanley, Ontario, into the western basin during August 1992, and into Hamilton Harbour in September 1992. Unless otherwise stated, 'top' refers to sites where transplanted mussels were placed 3.5 m under to surface of the water, and 'bottom' refers to sites where transplanted mussels were placed 1.0 m off the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	**Cd accumulation (μg Cd/g DW tissue/d) (mean ± 1 S.D.)**
E13 top	0.26 ± 0.04
E13 bottom	0.16 ± 0.04
Pelee Island top	0.08 ± 0.03
Pelee Island bottom	0.07 ± 0.04
Point Pelee top	0.11 ± 0.03
Point Pelee bottom	0.06 ± 0.01
E10 top	0.10 ± 0.02
E10 bottom	0.14 ± 0.04
H top	0.22 ± 0.04
H bottom	0.23 ± 0.07
Hamilton Harbour top (1 m below the surface)	-0.03 ± 0.01
Hamilton Harbour bottom (8.5 m below the surface)	-0.03 ± 0.01

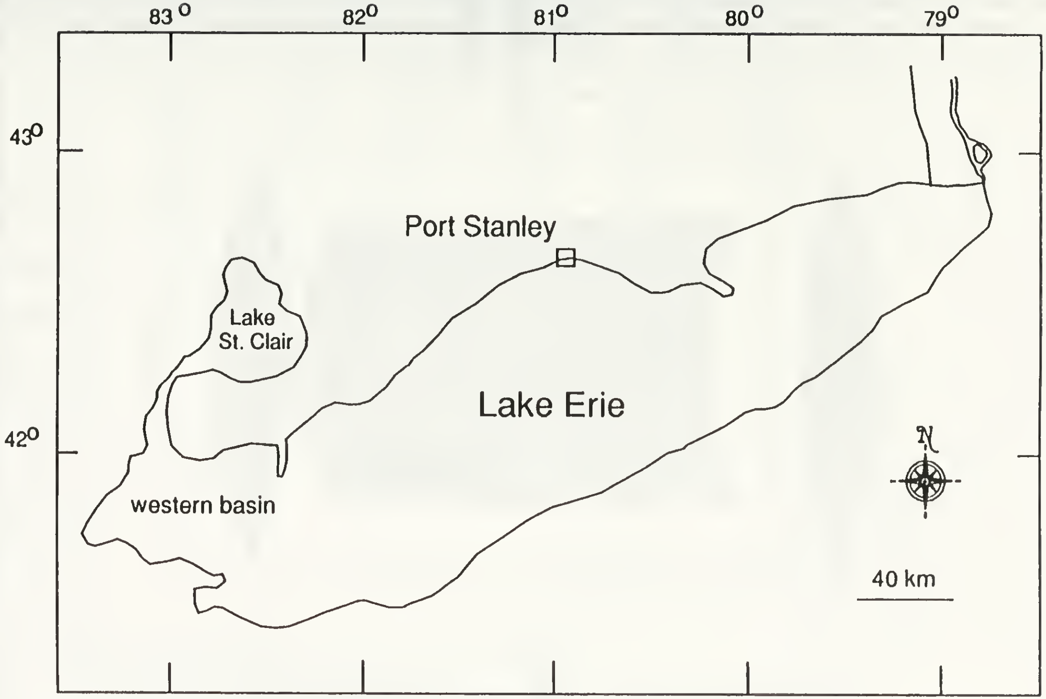


Figure 3.1. Map of Lake Erie showing location of initial mussel samples (Port Stanley, Ontario) used for transplanting into the western basin of Lake Erie, and Hamilton Harbour.

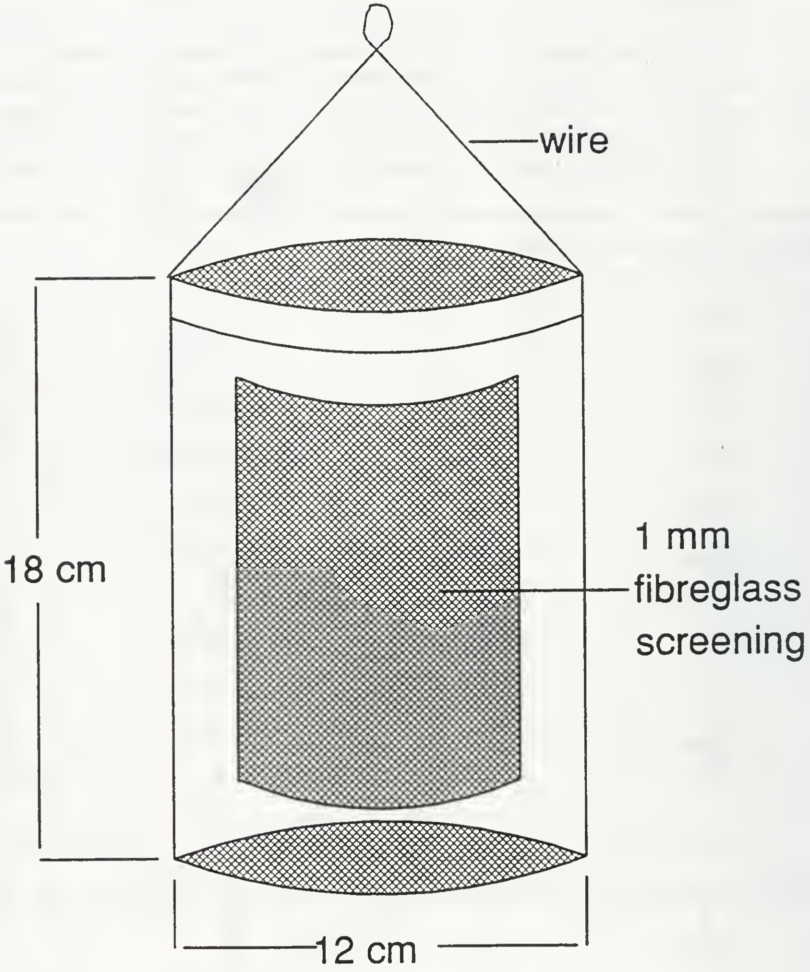


Figure 3.2. Cage design used to contain zebra mussels transplanted from Port Stanley, Ontario, into the western basin of Lake Erie (August 1992) and Hamilton Harbour (September-November 1992). Cage material is plexiglas and fibreglass screening.

Figure 3.3. PCB tissue levels (lipid corrected) for zebra mussel samples from Port Stanley, Ontario (initial), and after transplanting into the western basin of Lake Erie, August 1992. 'Top' refers to sites where transplanted mussels were placed 3.5 m under the surface of the water, and 'bottom' refers to sites where transplanted mussels were placed 1 m off the bottom of the lake. Refer to figure 2.1 and table 2.1 for specific locations and loran co-ordinates. Values shown are means ± 1 S.D.

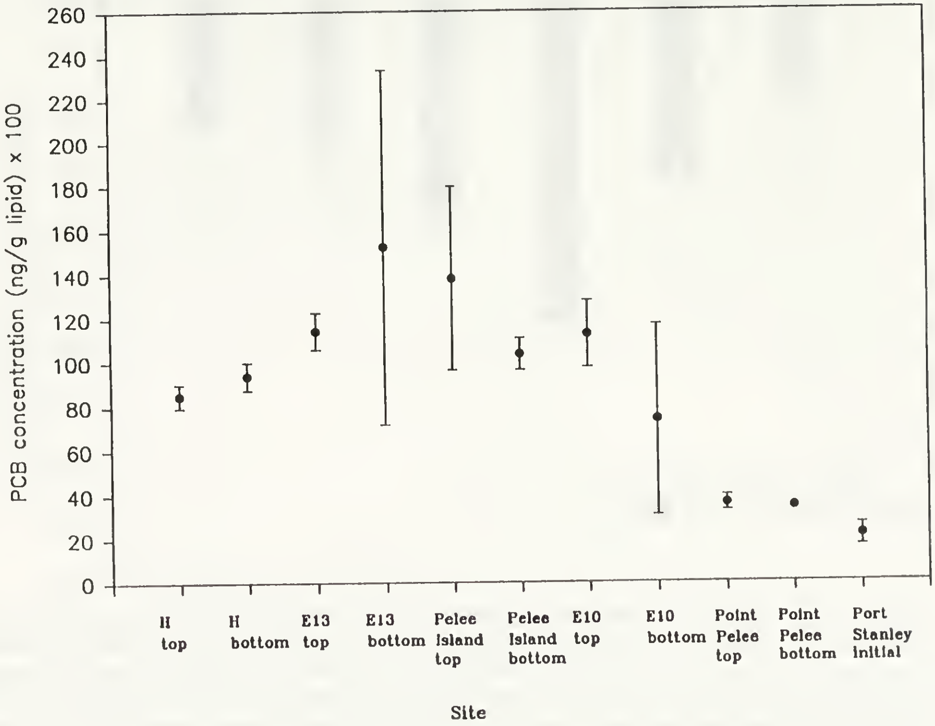


Figure 3.4. PCB tissue levels (lipid corrected) for zebra mussel samples transplanted into the western basin of Lake Erie from Port Stanley, Ontario, and for western basin of Lake Erie *in situ* mussels removed from coast guard buoys and chains. 'Top' refers to sites 3.5 m under the surface of the water, and 'bottom' refers to sites 1 m off the bottom of the lake. Refer to figure 2.1 and table 2.1 for specific locations and loran co-ordinates. Values shown are means ± 1 S.D.

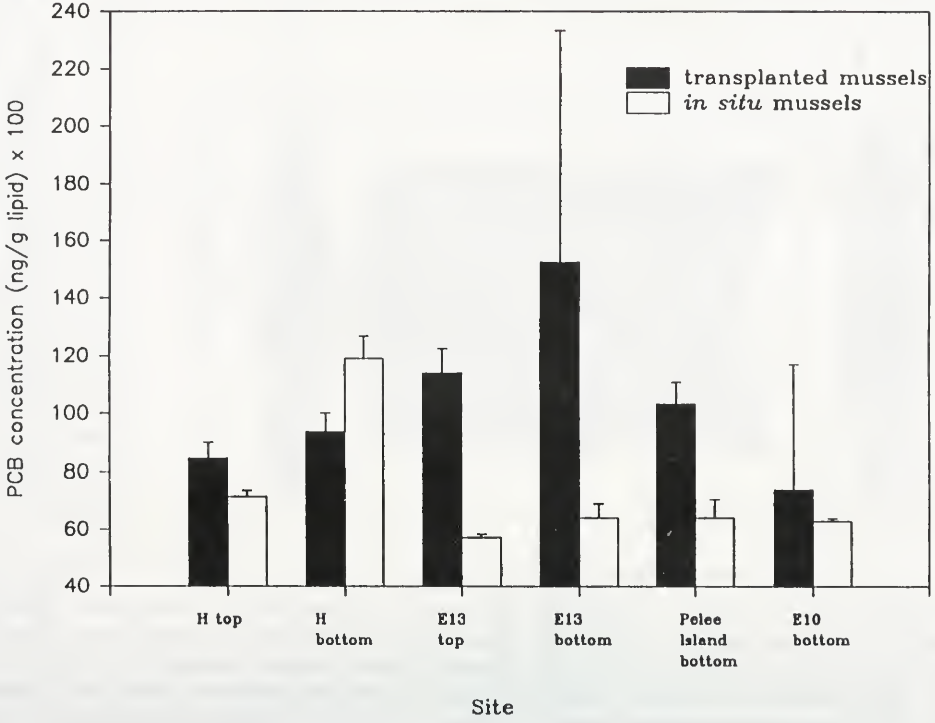


Figure 3.5. Cd tissue levels for zebra mussel samples from Port Stanley, Ontario (initial), and after transplanting into the western basin of Lake Erie, August 1992. 'Top' refers to sites where transplanted mussels were placed 3.5 m under the surface of the water, and 'bottom' refers to sites where transplanted mussels were placed 1 m off the bottom of the lake. Refer to figure 2.1 and table 2.1 for specific locations and loran co-ordinates. Values shown are means ± 1 S.D.

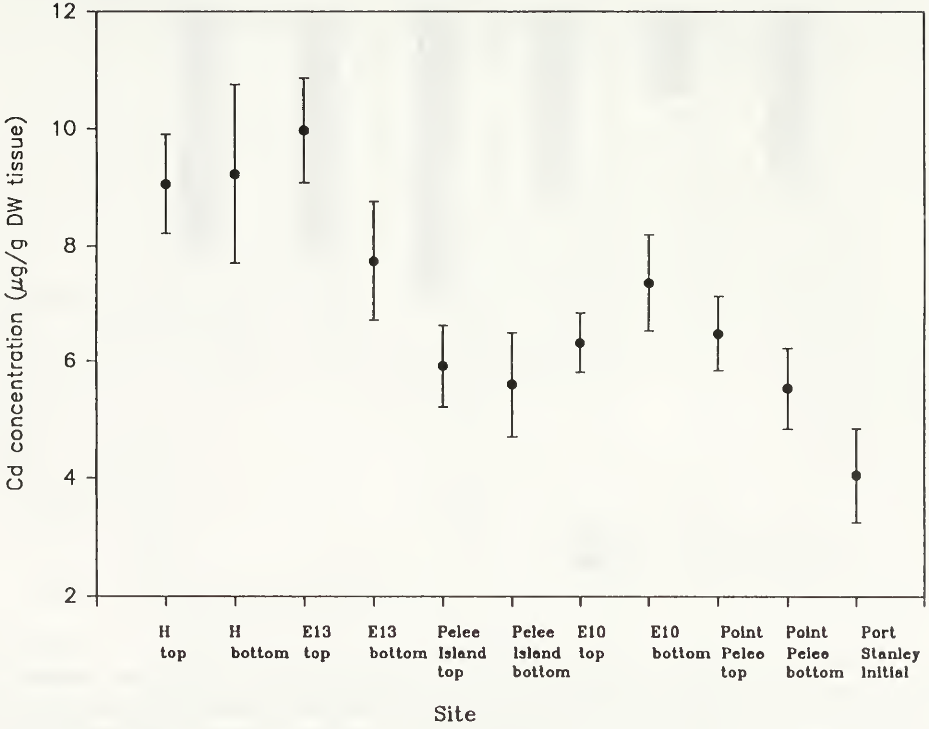


Figure 3.6. Cd tissue levels for zebra mussel samples transplanted into the western basin of Lake Erie from Port Stanley, Ontario, and western basin of Lake Erie *in situ* zebra mussels removed from coast guard buoys and chains. 'Top' refers to sites 3.5 m under the surface of the water, and 'bottom' refers to sites 1 m off the bottom of the lake. Refer to figure 2.1 and table 2.1 for specific locations and loran co-ordinates. Values shown are means ± 1 S.D.

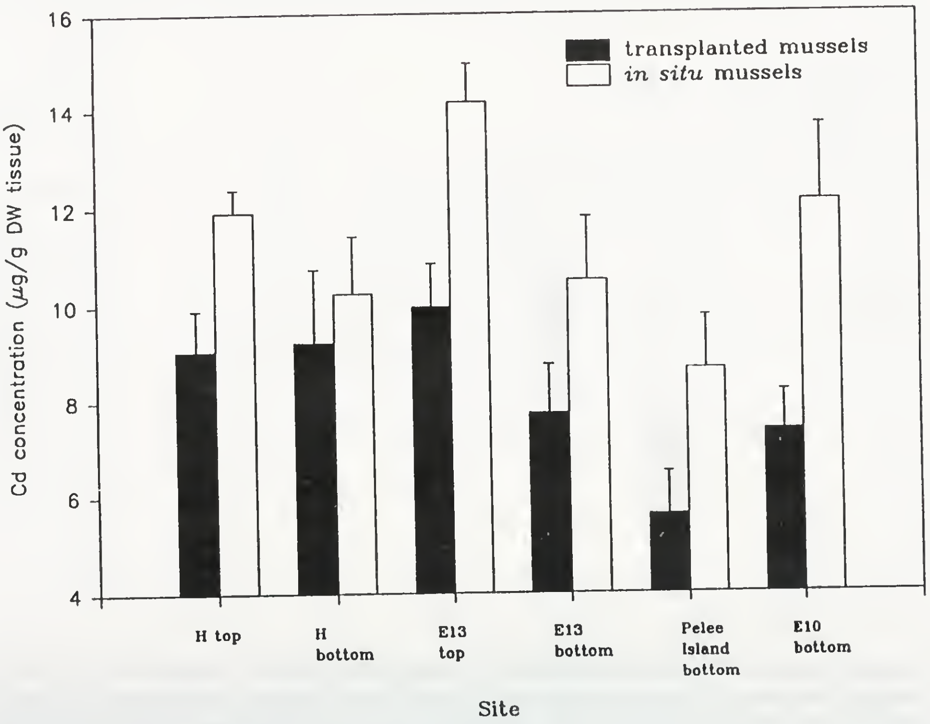
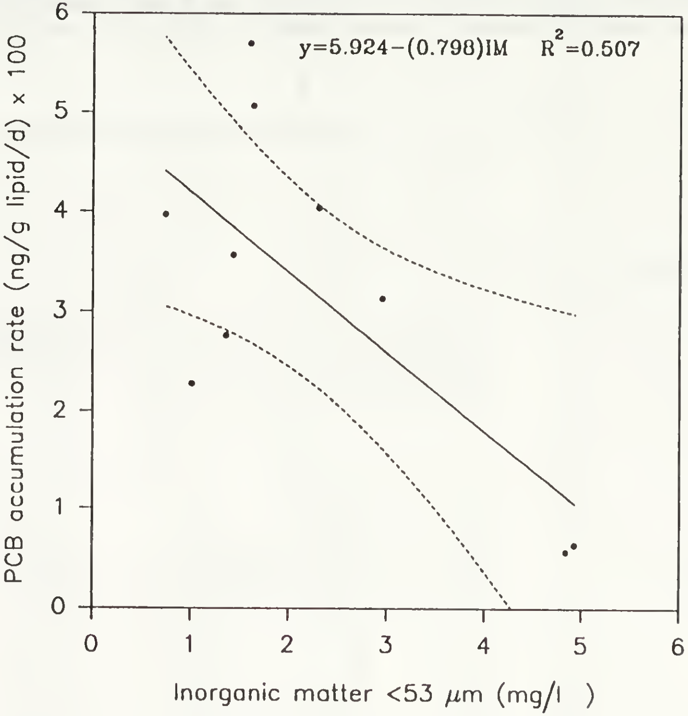
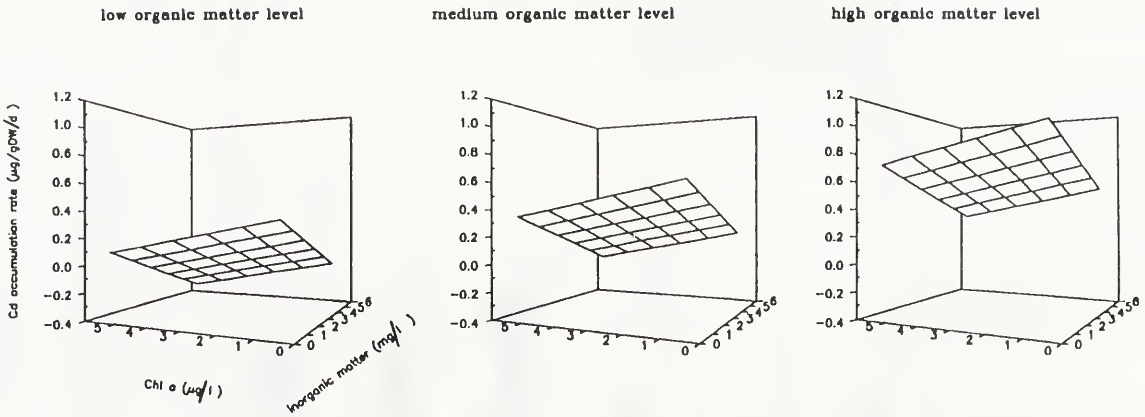


Figure 3.7. Regression of PCB accumulation rate of zebra mussels transplanted from Port Stanley, Ontario, to the western basin of Lake Erie versus suspended inorganic matter concetration <53 μm (IM) (N=10, bands are 95% confidence intervals).





$y = 0.15 - (0.08)IM + (0.51)OM - (0.05)chl\ a \qquad R^2 = 0.796 \qquad N = 10$

Figure 3.8. Cd accumulation rate model of zebra mussels transplanted from Port Stanley, Ontario, into the western basin of Lake Erie. Factors in the model are suspended inorganic matter concentration (<53 µm) (IM), chlorophyll a concentration (chl a), and suspended organic matter (concentration (<53 µm) (OM). The diagrams were generated by holding organic matter concentration constant at a low, medium, and high level in the equation, while inputing various data for the other two factors at each of these instances.

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Appendix I. Acid cleaning procedures for all mason jars and tissue sample vials.

1. Soaked in a concentrated solution of Sparkleen™ detergent for a minimum of 24 h.
2. Thoroughly cleaned with soap and water and rinsed with deionized water.
3. Soaked in a 10% solution of hydrochloric acid for 24 h.
4. Rinsed several times with deionized water.
5. Air-dried and then capped until used

Appendix II. Total dry weight of trap material in sedimentation traps and control traps for all sites in the western basin of Lake Erie, August 1992. 'Top' refers to sites where sampling apparati were placed 3 m under the surface of the water, and 'bottom' refers to sites where sampling apparati were placed 1.5 m off the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

Site	**weight of sediment in sedimentat traps (g DW) (mean ± 1 S.D.)**	**weight of sediment in control traps (g DW) (mean ± 1 S.D.)**
E13 top	17.2 ± 1.3	27.4 ± 4.0
E13 bottom	31.6 ± 8.2	24.2 ± 8.0
Pelee Island top	16.0 ± 1.3	15.4 ± 0.9
Pelee Island bottom	20.2 ± 1.8	19.5 ± 1.7
Point Pelee top	28.5 ± 1.7	25.4 ± 1.6
Point Pelee bottom	39.3 ± 4.1	36.5 ± 3.9
E10 top	20.7 ± 7.3	30.9 ± 3.6
E10 bottom	35.4 ± 0.5	34.6 ± 7.4
H top	13.8 ± 3.5	16.8 ± 4.5
H bottom	16.2 ± 2.6	13.7 ± 1.5

Appendix III. PCB concentrations in the trap material for sedimentation traps and control traps for all sites in the western basin of Lake Erie, August 1992. 'Top' refers to sites where sampling apparati were placed 3 m under the surface of the water, and 'bottom' refers to sites where sampling apparati were placed 1.5 m off the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

Site	PCB concentration for trap material in sediment traps (μg PCBs/g DW trap material) (mean ± 1 S.D.)	PCB concentration for trap material in control traps (μg PCBs/g DW trap material) (mean ± 1 S.D.)
E13 top	0.10 ± 0.01	0.14 ± 0.01
E13 bottom	0.17 ± 0.03	0.14 ± 0.05
Pelee Island top	0.16 ± 0.11	0.26 ± 0.15
Pelee Island bottom	0.14 ± 0.11	0.25 ± 0.16
Point Pelee top	0.10 ± 0.02	0.09 ± 0.01
Point Pelee bottom	0.08 ± 0.01	0.09 ± 0.00
E10 top	0.12 ± 0.04	0.13 ± 0.01
E10 bottom	0.11 ± 0.01	0.14 ± 0.01
H top	0.19 ± 0.07	0.19 ± 0.05
H bottom	0.17 ± 0.01	0.14 ± 0.04

Appendix IV. Cd concentrations in trap material in sedimentation traps and in control traps for all sites in the western basin of Lake Erie, August 1992. 'Top' refers to sites where sampling apparati were place 3 m under the surface of the water, and 'bottom' refers to sites where sampling apparati were placed 1.5 m off the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

Site	**Cd concentration for trap material in sediment traps (μg Cd/g DW trap material) (mean ± 1 S.D.)**	**Cd concentration for trap material in control traps (μg Cd/ g DW trap material) (mean ± 1 S.D.)**
E13 top	0.88 ± 0.02	0.81 ± 0.18
E13 bottom	1.03 ± 0.25	0.85 ± 0.04
Pelee Island top	0.55 ± 0.10	0.37 ± 0.13
Pelee Island bottom	0.48 ± 0.27	0.82 ± 0.23
Point Pelee top	0.88 ± 0.07	0.77 ± 0.07
Point Pelee bottom	0.88 ± 0.04	0.75 ± 0.04
E10 top	0.98 ± 0.31	0.66 ± 0.01
E10 bottom	0.75 ± 0.19	0.79 ± 0.03
H top	1.11 ± 0.34	0.51 ± 0.13
H bottom	0.51 ± 0.25	0.76 ± 0.08

Appendix V. General linear model procedure for biodeposition rate (DEPR) versus inorganic matter <53 μm (IM).

```
Dependent Variable: DEPR
                           Sum of        Mean
Source              DF      Squares        Square  F Value   Pr > F

Model                1   0.17284798   0.17284798    4.33     0.0710

Error                8   0.31918842   0.03989855

Corrected Total      9   0.49203640

            R-Square        C.V.       Root MSE        MDEPR Mean

            0.351291     67.57315      0.199746       0.29560000


Source              DF     Type I SS   Mean Square  F Value    Pr > F

IM                   1   0.17284798   0.17284798    4.33       0.0710

Source              DF   Type III SS   Mean Square  F Value    Pr > F

IM                   1   0.17284798   0.17284798    4.33       0.0710
```

Appendix VI. Environmental factor data collected over the study period for all sites in the western basin of Lake Erie, August 1992. 'Top' refers to sites where data were collected 3 m under the surface of the water, and bottom refers to sites where data were collected 1.5 m off the bottom of the Lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

Site	**chlorophyll a (μg/l)**	**suspended inorganic matter <53 μm (mg/l) (mean ± 1 S.D.)**	**suspended organic matter <53 μm (mg/l) (mean ± 1 S.D.)**
E13 top	3.84 ± 0.82	2.31 ± 1.30	1.23 ± 0.23
E13 bottom	3.66 ± 0.99	1.61 ± 0.29	0.95 ± 0.16
Pelee Island top	2.40 ± 1.16	1.64 ± 1.49	0.65 ± 0.29
Pelee Island bottom	1.75 ± 0.91	1.43 ± 0.28	0.67 ± 0.04
Point Pelee top	5.15 ± 1.71	4.93 ± 2.04	1.52 ± 0.23
Point Pelee bottom	3.62 ± 0.86	4.84 ± 2.84	1.37 ± 0.50
E10 top	1.62 ± 0.52	0.74 ± 0.41	0.55 ± 0.14
E10 bottom	1.19 ± 0.30	1.01 ± 0.67	0.58 ± 0.10
H top	2.88 ± 1.68	1.36 ± 0.25	1.05 ± 0.05
H bottom	1.40 ± 0.42	2.95 ± 1.22	1.02 ± 0.24

Appendix VII. Biodeposition rate of PCBs versus inorganic matter <53 μm, organic matter <53 μm, and their interaction.

```
Dependent Variable: DEP
                            Sum of         Mean
Source              DF     Squares        Square   F Value   Pr > F

Model                3   0.00659616   0.00219872     5.21    0.0416

Error                6   0.00253447   0.00042241

Corrected Total      9   0.00913063

             R-Square         C.V.      Root MSE         MDEP Mean

             0.722421     46.97746      0.020553        0.04375000


Source              DF    Type I SS   Mean Square  F Value    Pr > F

IM                   1   0.00098591   0.00098591     2.33     0.1774
OM                   1   0.00284565   0.00284565     6.74     0.0409
IMOM                 1   0.00276460   0.00276460     6.54     0.0430

Source              DF  Type III SS   Mean Square  F Value    Pr > F

IM                   1   0.00096598   0.00096598     2.29     0.1812
OM                   1   0.00532708   0.00532708    12.61     0.0121
IMOM                 1   0.00276460   0.00276460     6.54     0.0430
```

Appendix VIII. Cd biodeposition versus inorganic matter <53 μm (IM), chlorophyll a (CHL), and their interaction.

```
Dependent Variable: DEP
                           Sum of         Mean
Source             DF      Squares        Square     F Value    Pr > F

Model               3    0.50770710    0.16923570     18.67     0.0019

Error               6    0.05437540    0.00906257

Corrected Total     9    0.56208250

          R-Square         C.V.       Root MSE       MDEP Mean

          0.903261       31.47025     0.095198      0.30250000


Dependent Variable: MDEP

Source             DF     Type I SS    Mean Square  F Value     Pr > F

IM                 1     0.22865550    0.22865550    25.23      0.0024
CHL                1     0.06733804    0.06733804     7.43      0.0344
IMCHL              1     0.21171356    0.21171356    23.36      0.0029

Source             DF   Type III SS    Mean Square  F Value     Pr > F

IM                 1     0.04024010    0.04024010     4.44      0.0797
CHL                1     0.27671007    0.27671007    30.53      0.0015
IMCHL              1     0.21171356    0.21171356    23.36      0.0029
```

Appendix IX. Average mussel shell length for PCB transplant samples in the western basin of Lake Erie, and Hamilton Harbour. Unless otherwise stated, 'top' refers to sites where transplanted mussels were placed 3.5 m under the surface of the water, and 'bottom' refers to sites where transplanted mussels were placed 1 m off the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	**mean length (μm) ± 1 S.D.**
E13 top	12297.8 ± 467.0
E13 bottom	13377.5 ± 622.5
Pelee Island top	12653.3 ± 431.3
Pelee Island bottom	13963.7 ± 497.6
Point Pelee top	14119.3 ± 388.1
Point Pelee bottom	13891.8 ± 555.5
E10 top	13248.0 ± 419.3
E10 bottom	14474.5 ± 428.5
H top	14692.3 ± 438.9
H bottom	13251.3 ± 774.6
Hamilton Harbour top (1 m below the surface)	14281.7 ± 674.2
Hamilton Harbour bottom (8.5 m below the surface)	14287.0 ± 299.8

Appendix X. Percent lipid content of all mussel samples (transplanted and *in situ*) from the western basin of Lake Erie, Port Stanley initials, and Hamilton Harbour. Unless otherwise stated, 'top' refers to sites where mussels were placed or removed approximately 3.5 m under the surface of the water, and 'bottom' refers to sites where mussels were placed or removed approximately 1 m off the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	**percent lipid content (mean ± 1 SD)**
E13 top	0.80 ± 0.08
E13 bottom	0.61 ± 0.20
Pelee Island top	0.95 ± 0.33
Pelee Island bottom	0.52 ± 0.15
Point Pelee top	1.28 ± 0.49
Point Pelee bottom	0.82 ± 0.13
E10 top	0.80 ± 0.18
E10 bottom	0.66 ± 0.23
H top	0.79 ± 0.21
H bottom	0.88 ± 0.09
Western Basin initial Port Stanley	0.63 ± 0.10
H top *in situ*	2.93 ± 0.12
H bottom *in situ*	1.25 ± 0.05
E13 top *in situ*	2.37 ± 0.12
E13 bottom *in situ*	1.20 ± 0.20
E bottom *in situ*	0.40 ± 0.08
E10 bottom *in situ*	0.55 ± 0.08
Hamilton Harbour top (1 m below the surface)	1.10 ± 0.10
Hamilton Harbour bottom (8.5 m below the surface)	0.93 ± 0.06
Hamilton Harbour initial Port Stanley	0.44 ± 0.04

Appendix XI. Average mussel shell length for Cd transplant samples in the western basin of Lake Erie, and Hamilton Harbour. Unless otherwise stated, 'top' refers to sites where transplanted mussels were placed 3.5 m under the surface of the water, and 'bottom' refers to sites where transplanted mussels were placed 1 m off the bottom of the lake. Refer to Figure 2.1 and Table 2.1 for specific locations and Loran coordinates.

site	**mean length (μm) ± 1 S.D.**
E13 top	13697.8 ± 203.1
E13 bottom	14186.3 ± 851.0
Pelee Island top	13927.8 ± 937.9
Pelee Island bottom	13826.3 ± 1078.7
Point Pelee top	13483.0 ± 806.1
Point Pelee bottom	13889.5 ± 834.8
E10 top	12172.0 ± 1968.6
E10 bottom	13896.0 ± 160.1
H top	13861.5 ± 909.4
H bottom	13695.0 ± 1307.4
Hamilton Harbour top (1 m below the surface)	13377.5 ± 711.8
Hamilton Harbour bottom (1 m below the surface)	13705.5 ± 441.7

Appendix XII. PCB accumulation rate (ACM) versus inorganic matter <53 μm (IM).

```
Dependent Variable: ACM
                         Sum of          Mean
Source            DF    Squares         Square  F Value   Pr > F

Model              1   13.02394534   13.02394534    8.23    0.0209

Error              8   12.65985156    1.58248144

Corrected Total    9   25.68379690

             R-Square        C.V.      Root MSE        MACM Mean

             0.507088      30.66642    1.257967       4.10210000


Source            DF     Type I SS    Mean Square  F Value    Pr > F

IM                 1    13.02394534   13.02394534    8.23      0.0209

Source            DF   Type III SS    Mean Square  F Value    Pr > F

IM                 1    13.02394534   13.02394534    8.23      0.0209
```

Appendix XIII. Cd accumulation rate (ACM) versus inorganic matter <53 μm (IM), organic matter <53 μm (OM), and chlorophyll a (CHL).

```
Dependent Variable: MACM
                              Sum of          Mean
Source             DF         Squares         Square   F Value    Pr > F

Model               3      0.03560271     0.01186757     7.81      0.0171

Error               6      0.00911796     0.00151966

Corrected Total     9      0.04472068

             R-Square          C.V.       Root MSE         MACM Mean

             0.796113       12.24643      0.038983         0.31832000


Dependent Variable: ACM

Source             DF      Type I SS    Mean Square   F Value     Pr > F

IM                 1      0.00112471     0.00112471     0.74      0.4227
OM                 1      0.02436431     0.02436431    16.03      0.0071
CHL                1      0.01011369     0.01011369     6.66      0.0418

Source             DF     Type III SS   Mean Square   F Value     Pr > F

IM                 1      0.02989464     0.02989464    19.67      0.0044
OM                 1      0.03292096     0.03292096    21.66      0.0035
CHL                1      0.01011369     0.01011369     6.66      0.0418
```